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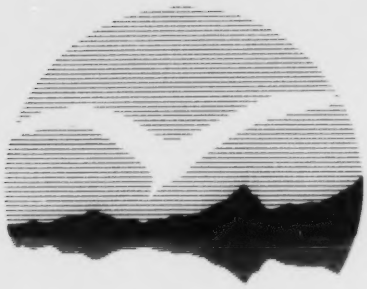

WATER
RESOURCES
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OFF-FARM SEDIMENT IMPACTS

in the

SASKATCHEWAN RIVER BASIN

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in the

SASKATCHEWAN RIVER BASIN

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by

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for

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EXECUTIVE SUMMARY

1. This report is an assessment of the nature, extent and effects of off-farm sediment impacts in the Saskatchewan River system as they relate to streamborne sediments and associated contaminants, i.e. how much of the sediments originate from farmland and what are the impacts of this farmland sediment on instream and off-stream uses of water in the river basin.

2. Studies in the USA have shown farmland erosion to be a major source of sediments in streams with serious impacts: high levels of suspended solids adversely affect fish performance, in-stream and off-stream engineering activity; increased rates of sediment accumulation spoil fish habitat, infill channels and produce high dredging costs in ditches, canals and reservoirs: and contaminants that are bound to off-farm sediment (nutrients, heavy metals and organic compounds) have impacts on aquatic life.

3. Similar studies in Canada have been restricted to southern Ontario and have revealed much less serious impacts: though farmland erosion rates were quite high (1-10 t/ha/yr), sediment yields from cultivated basins were an order of magnitude smaller (10-100 $t/km^2/yr$) and were much smaller than in the USA; heavy metals (from fertilisers) bound to sediments were negligible; and though pesticides occasionally showed up in stream water no deleterious effects on fish were noted. It was determined, however, that agricultural land accounted for the majority of the diffuse load of phosphorus delivered to the Great Lakes, and most of this was bound to sediment.

4. Measurement of soil erosion on farmland in the Prairies indicates highly variable rates up to 70 t/ha/yr, but net losses from fields are much smaller, though as high as 20 t/ha/yr in some cases; sediment yields in small stream basins were again much lower (1-50 $t/km^2/yr$), and slightly smaller than in southern Ontario. Sediment yields increased on larger streams and are typically in the range 50-100 $t/km^2/yr$ at the downstream ends of the Red Deer, South and North Saskatchewan rivers. It is inferred that this increment in sediment production along the channel network is primarily the result of erosion of channel banks and gullying of valley slopes. This conclusion implies that, even if the effects of farmland sediment are noticeable in small streams, they will be masked in the larger rivers because of the amount of "natural" non-farm sediment.

5. Suspended sediment concentrations are generally low in small streams (median open-water season levels of 10-50 mg/L) and become higher in downstream reaches of mainstem rivers (median levels greater than 100 mg/L). Some rivers have levels of 1,000 mg/L during short periods (less than 1% of the time). These levels are still relatively low most of the time, however, and no problems are known with in-stream use of water arising directly from high turbidity levels.

6. The disparity between on-farm soil erosion rates and in-stream sediment yields implies accumulation of sediment in the watersheds upstream of the sampling stations. Yet there is little information to indicate that sedimentation has proved a serious problem in small watercourses in the system. Whether this is because in-stream sedimentation is in fact not a real problem (farmland sediment simply accumulating in low-slope areas before reaching streams) or whether it is due to the lack of pertinent data is hard to assess. Clearing of sediment from roadside ditches and irrigation canals is certainly a recurrent task, but is often simply included in general operating budgets and not clearly itemized.

7. Nutrient levels in the mainstem water bodies (where little of the sediment results from farmland erosion) seem to be close to background levels in most places and most of the time: phosphorus levels in the sediment settled in Lake Diefenbaker, for example, are comparable with those in surface soils which, in turn, are not appreciably higher than those in subsurface sediment. Occasional eutrophication does occur on the main stems but this is largely a summer phenomenon downstream of sewage treatment plants. Smaller streams and lakes appear to show much greater eutrophication, however, with high nutrient levels directly related to farmland runoff and soil erosion; the latter, involving selective removal of organic matter and fines, tends to be enriched in sediment-associated phosphorus.

8. Data for concentrations of heavy metals in stream sediments are generally restricted to mainstem sites. They confirm that many of the metals are preferentially bonded to sediments (rather than being dissolved in the water) but indicate levels that are generally comparable with natural concentrations in cultivated and non-cultivated surface soils and subsurface sediments. The exception is mercury, but the source of mercury contamination is difficult to determine. In some cases it is clear that the source is chloralkali plants, but levels in fish (the data being much more abundant than for sediments) indicate a fairly widespread contamination. Whether this is indicative of a ubiquitous source (such as fungicide previously applied on farmland) or whether it simply reflects the movement of fish is unclear. Few metal data are available for small streams, but given the findings above, it is not expected that levels would be high.

9. Several organic contaminants resulting from pesticides have been detected in the basin, though whether these levels are harmful is uncertain. Levels in fish (generally higher than in sediments) are regarded as safe. Most of the data are from mainstem sites where the sediment is largely from non-farm sources. In smaller streams with a greater proportion of sediment from farmland (but where no data appear to exist) levels may be much higher. On the other hand, such small waterbodies are much less intensively used by man.

10. This review of data for the Saskatchewan River basin indicates that off-farm sediment impacts here are less than found in southern Ontario. Whether this is a fair comparison, however, or whether the conclusion is restricted to the particular sets of sample data, is a separate question. The southern Ontario data were collected (as part of a special pollution investigation) in small drainage basins (less than 80 km^2) in which most of the in-stream sediment originated from farmland erosion. The Saskatchewan River basin data have been collected primarily at mainstem sites where the drainage area is generally thousands of square kilometres. In these larger basins, a great deal of natural sediment is picked up through bank and slope erosion along the river, and this sediment masks the sediment attributes of tributary streams.

11. There is evidence to indicate that nutrient enrichment of streams and lakes due to farmland erosion is much more pronounced in small watercourses. It seems likely, therefore, that levels of other sediment-related contaminants, such as pesticide residues, would be higher in these streams. Little or no data are available at this scale. Heavy metal contamination will probably be little different in the small basins because farmland soils do not seem to be heavily enriched compared to natural sediments.

12. The suggestion that farmland contaminants may be more apparent in sediments of smaller streams should not be misinterpreted as indicating that sediment pollution, overall, is greatest in smaller basins. On the contrary, sediment yields in small basins appear to be smaller than in large basins for reasons already given. This means that sediment pollution itself (in terms of high turbidity and sedimentation of watercourses) is unlikely to be worse than in the main stems. The suggestion is. rather, that small watersheds, though yielding less sediment, probably produce a much greater <u>proportion</u> of that sediment from farmland erosion. Thus contaminants associated with eroded farmland soil may be present in fluvial sediments in higher concentrations.

13. Whether the inferred pollution of smaller watercourses in the basin is a problem will depend on what uses are made of this water now and in the future. It would seem worthwhile, however, to direct some portion of future sediment-quality investigations to representative small watersheds in order to ascertain data on the topic, as in the case of the PLUARG studies in southern Ontario.

14. Various evidence indicates that farmland erosion and its off-farm costs are much more severe in other parts of the Prairies, especially in the Peace River district.

15. Recommendations are made for further analysis of existing data as well as new field monitoring of small catchments.

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1. INTRODUCTION

1.1 The problem

a. Terms of reference

The terms of reference of this work were : "to prepare a report which contains an assessment of the nature, extent and effects of off-farm impacts in the Saskatchewan River system as they relate to stream borne sediments and what they carry."

Its purpose is to assemble information that currently exists on the topic; to determine the extent to which this presently-available information can be used to address off-farm sediment issues; to indicate the conclusions that emerge from this data file where adequate information exists; and to recommend future work in cases where insufficient data are available.

b. Scope of the study

The scope of this study includes both:

- the transfer (by both wind and water) of sediment (and contaminants bound up with the sediment) off-farm to the drainage system;
- and the scavenging by sediment in the drainage system of farmland contaminants from atmospheric fallout and those dissolved in water draining from farmlands.

In practice, however, it is often very difficult to separate these two processes. The point is emphasized, however, because the terms of reference do not include contaminants that are dissolved in farmland runoff and that stay dissolved as that water drains downstream through the drainage system. The report is concerned with streamborne sediments and the farming contaminants that they may adsorb from the water phase or desorb to the water phase.

The term "drainage system" is used here in a broad sense to include man-made watercourses (drainage ditches, roadside ditches and irrigation canals) as well as natural streams and rivers, and lakes and man-made reservoirs on these watercourses.

It should be emphasized that the project is not a study of the impact of river sediments on water use in general, but a study of these problems only as they relate to sediments and sediment-bound contaminants that come from farmland. Partitioning sediment to agricultural and non-farming sources is no simple matter, however, as will appear throughout this report.

The study must really address five topics:

i. How much sediment in the Saskatchewan River system comes from farmland? How much contaminant bound up with sediment in the stream network comes from farmland? What is the regional pattern?

ii. What percentage of the sediment in the Saskatchewan river system is from farmland? What percentage of sediment-bound contaminants in the stream network is from farmland? What is the regional pattern?

iii. What are the impacts of sediment and sediment-bound contaminants (from all sources) in the Saskatchewan River system. What is the regional pattern?

iv. What proportion of the impacts is attributable to sediment and sediment-related contaminants from farmland? What is the regional pattern?

v. How important are these impacts in the different areas affected?

These five sets of questions need to be considered in each general category of potential impacts outlined below.

1.2 Potential impacts

Review of the general literature dealing with "stream pollution" in agricultural areas indicates three main groupings of potential impacts of farming on stream sediments. These are:

a. those associated with the increased concentration of suspended sediment in streams and other parts of the natural drainage network, e.g. impacts of high turbidity on fish performance, damage to hydraulic machinery due to suspended sediment, costs of removal of sediment from water prior to off-stream use, etc.

b. those associated with increased rates of sediment accumulation in waterways, e.g. deterioration of fish habitat, increased flooding through channel infilling, costs of removal of sediment from ditches, canals, reservoirs etc.

c. those associated with increased concentration of contaminants in suspended and bottom sediments in the channel network, e.g. eutrophication from washing of nutrients into streams and the role played by sediment in controlling the bioavailability of these nutrients; and the role of sediment in the bioavailability of metals and pesticides from farmland and the impact on aquatic life, etc.

1.3 The American experience

The off-farm impacts of eroding soils have recently attracted considerable attention in the USA (Clark et al., 1985). They note that in 1977 the US Dept. of Agriculture conducted its first National Resource Inventory and concluded that agricultural lands accounted for 69% of all water-caused erosion in the USA. They provided estimates of the various off-farm impacts (Table 1.1), though they emphasize the difficulties involved not only in costing the impacts of sediment and its associated contaminants, but also assessing the true share applicable to farmland. It was found impossible, in fact, to cost the biological impacts (primarily on fish) of farmland erosion.

1.4 The Canadian scene

Nothing comparable with the report by Clark et al. exists for Canada. A preliminary economic assessment of agricultural land degradation in Atlantic and Central Canada and southern British Columbia was prepared for Agriculture Canada (DCH-LRRI, 1986) which included comments on off-farm impacts in these regions. The report made some rough estimates of the potential off-farm economic impacts based on extrapolation from the Clark report and unpublished estimates for Ontario. In Ontario alone (Table 1.2), the annual off-farm impacts were assessed at more than $90 million, primarily as a result of recreational fisheries losses (ESCC, 1983). Impacts on recreational fishing in the Prairies would be anticipated to be much smaller.

Attention to the impacts of farmland erosion has been most evident in Ontario with the establishment of a 5-year study group known as PLUARG (Pollution from Land Use Activities Reference Group) in 1972 by the International Joint Commission to determine the extent to which runoff from the land is involved in pollution in the Great Lakes. Among the findings of that investigation (Coote, 1980) were:

- agricultural land accounted for 60% of the diffuse load of phosphorus (much of it bound to sediment) delivered to the Great Lakes;
- overall fertiliser contributions of lead, mercury, arsenic, cadmium and selenium (bound to sediment) to streams were negligible;
- while pesticides occasionally showed up in stream water no deleterious effect on fish was noted;
- PCBs were found in all agricultural watersheds (presumably from industrial sources through the atmosphere) and sorbed to farmland sediment:

- nitrates were found in some streams at levels greater than that deemed safe for drinking (10 mg/L as N) and were therefore a concern to farmers and others who use stream water for domestic use.

The latter finding is not directly relevant in the present review because nitrates move from the land into and through the stream essentially in solution, and have minimal involvement with sediment.

An important component of the PLUARG work, in the present context, was the study of fluvial sediments by Wall et al. (1982), sampling suspended sediments discharged from 11 small (<60 km^2) agricultural watersheds. The 1975-77 specific sediment yields in these basins ranged from 10 to 100 t/km^2/yr, apparently quite strongly influenced by the percentage row crop in the watershed and the % clay in the surface soil. The study also examined sediment sources within the watersheds (Table 1.3) concluding that the bulk of all streamborne sediment originated from cropland. Sediment yields in streams were only 10% of sediment losses estimated from fields, the latter being estimated at 1-10 t/ha/yr.

A post-PLUARG study by Dickinson and Pall (1982) in five small subbasins of the Thames River watershed determined average sediment yields of 80 t/km^2/yr, but, using the GAMES sediment erosion model (Guelph model for evaluating the effects of Agricultural Management Systems on Erosion and Sedimentation), found rates at the scale of field-sized areas to be highly variable. Spring time soil losses ranged from 0 to 14 t/ha in the more rolling upland watersheds (Fig. 1.1).

1.5 Approach in this report

There are two basic approaches possible in this overview. The first is to identify parts of the Saskatchewan River network in which sediment supply from farmland is high, and then to assess what impacts, if any, have resulted from this excess farmland sediment. The second is to determine where problems have been identified as resulting from high sediment levels in the water system, and then to assess how much, if any, of this sediment impact is due to farming.

Both approaches are taken in the chapters that follow. Soil erosion rates on farmland within the Saskatchewan River basin are discussed, and compared with sediment yields in the stream network, in Chapter 2.

Chapter 3 examines problems arising from high suspended sediment concentrations in the drainage network of the area, and attempts to assess how much of this is due to farmland erosion.

Chapter 4 examines the impacts of increased rates of sedimentation along watercourses in the basin and, again,

attempts to assess how much results from farming.

Chapters 5 (nutrients), 6 (heavy metals) and 7 (organic compounds) look at problems associated with increased levels of contaminants in suspended and bottom sediments and attempt to establish the contribution to these problems from farmland erosion, runoff and irrigation return flows.

The conclusions are presented in Chapter 8; recommendations for additional work in order to more accurately identify the role of farming in some of the sediment-related issues in the Saskatchewan River basin are made in Chapter 9.

1.6 The Saskatchewan River basin: environmental setting

The environmental setting of the Saskatchewan River basin has already been summarized by Ashmore (1986). Certain key features need emphasizing in the context of the present report, especially in relation to the connectivity of the drainage network.

Meid and Schoenfeld (1989) report the gross drainage area of the Saskatchewan River at The Pas, Manitoba to be 347,000 km^2 . This figure does not include three large dead drainage basins (Mowchenko and Reid, 1983) which drain internally into lakes with no outlet: Maple Creek basin (13,755 km^2); Pakowki Lake (5,318 km^2) and Sounding/Eyehill drainage basin (25,152 km^2). More importantly, however, only part of the gross drainage area actually contributes runoff to the main stem under normal conditions. This is referred to as the "effective" drainage area: that portion of a basin which might be expected to entirely contribute runoff to the main stream during a flood with a return period of two years (Mowchenko and Reid, 1983). This area excludes marsh and slough areas and other natural storage areas which would prevent runoff from reaching the main stream in a year of average runoff.

Because of the rolling, hummocky terrain typical of much of the Prairies, a considerable part of the gross drainage area does not contribute runoff to the main stem under normal conditions. Meid and Schoenfeld (1989) in fact report an effective drainage area at The Pas of only 224,000 km^2, slightly less than two-thirds of the gross drainage area. Within the Plains proper (ignoring the Oldman above Brocket, the Bow above Calgary, the Red Deer above Red Deer and the North Saskatchewan above Edmonton), the effective area drops to less than 60 percent. This poor connectivity of much of the Prairie landscape with the mainstem drainage network must be borne in mind in any comparison of soil erosion rates on farmland with sediment yields in the streams and rivers.

The main streams of the basin, where they leave the Rocky Mountain foothills, become deeply incised into the Prairie landscape (sometimes more than 100 m). Undercutting of their

valley slopes is an obvious source of streamborne sediment, especially from weakly consolidated glacial deposits, but also from the relatively soft underlying pre-Quaternary sediments. The most spectacular example is along the lower Red Deer River (Alberta) where intense gullying of Cretaceous sediment produces annual erosion volumes that actually exceed the sediment load of the Red Deer monitored by WSC at Bindloss (Campbell, 1973).

The physical geography of the basin is summarized in the maps of relief, mean annual precipitation, mean annual runoff and natural vegetation given in Fig. 1.2 (Ashmore and Day, 1988). The area of lowest mean annual precipitation (300 mm), which peaks during June-July (Fig. 1.3), is centred on the confluence of the Red Deer and South Saskatchewan rivers, but, because of hot summers, most of the plains area of the basin has a mean annual runoff of less than 25 mm. Soil zones are strongly associated with the pattern of runoff (Fig. 1.4): Brown and Dark Brown soils in the drier area and more organic-rich Black and Gray Soils in the more moist areas.

The natural vegetation of the foothills is mixed aspen and pine or spruce woodland, giving way to aspen parkland in the northern part of the Plains and Prairie grassland in the southern Plains area (Ashmore, 1986). Through the plains, however, agriculture has replaced much of the natural vegetation, except in some riparian areas. Though rough pasture is found in some parts of the basin (the Cypress Hills and many of the valley slopes) most of the plains area is cropland, especially for cereals.

The shortage of soil moisture during the growing season led early settlers to introduce a summerfallow cropping system whereby only half of the crop land was cultivated each year in order to save enough moisture for a successful crop the following year (Anderson, 1975). Areas of summerfallow are generally seen as major sites of soil erosion: Coote (1983), for example, estimated average rates of water erosion (using the universal soil loss equation: see Section 2.4) of 38 t/ha/yr for fallow compared to only 19 t/ha/yr for row crops and 11 t/ha/yr for cereals and canola. Even today, summerfallow is estimated to occupy 24% of the cultivated land in Alberta and 38% in Saskatchewan (SSC, 1984).

The limited availability of natural soil moisture has also led to largescale irrigation in the southern part of the basin, especially in the Palliser Triangle (between the Red Deer River, the Rocky Mtn foothills and the USA border) where more than 4,000 km^2 of land are irrigated by 10,000 to 20,000 km of distribution canals and laterals (Joseph and Ongley, 1986), using water from the Bow and Oldman rivers. The huge extent of irrigation in Alberta compared to Saskatchewan is indicated by Figs. 1.5 and 1.6. Significant inter-basin transfers of water are involved in some of these schemes: diversion from the Bow through irrigation systems with return flows to the Red Deer and

Oldman rivers; and transfers out of the Waterton, the Belly and St. Mary rivers with return flows into the Oldman River downstream of Lethbridge.

The existence of dams (hydro power as well as irrigation) across the streams of the Saskatchewan River system obviously exerts a strong influence on the downstream change in sediment loadings and concentrations. A list of the major onstream storage reservoirs is provided by Ashmore (1986) and given in Table 1.4, together with their locations in Fig. 1.7. The only major river presently without a dam in its headwater reach is the Oldman River.

Information on sediment trapping by these dams has been found for only a few reservoirs. Yuzyk (1984), in a review of reservoir sedimentation across Canada, includes reference to several in the Saskatchewan River basin, and these are discussed more fully in Section 2.

Though precipitation on the plains peaks during summer (Fig. 1.3), most of the small plains streams produce maximum runoff in the spring as a result of snowmelt (the Battle R., for example, peaks in April). Others, with their headwaters in higher terrain, experience snowmelt peaks later, usually in June, as on the Oldman, Bow and Red Deer, while the North Saskatchewan River at Saskatchewan Crossing has its highest monthly flow in July. The effects on flow regulation (for hydro power and irrigation) are most apparent in the South Saskatchewan river system, spring and summer peaks being reduced while winter flows tend to be augmented.

It is important to recognize the major contribution in terms of runoff in the main stem rivers provided by the headwater areas as distinct from the farmlands of the plains. More than 70% of the mean annual flow of the South Saskatchewan River at Highway 41 (near the Alberta border) originates from the Bow upstream of Calgary and the Oldman River upstream of Brocket. Almost the same percentage of the Red Deer's runoff at Bindloss comes from upstream of Red Deer municipality. And 94% of the runoff of the North Saskatchewan River at Prince Albert is derived from upstream of Edmonton, 56% of it from upstream of Rocky Mountain House.

In summary, and without wishing to impose preconceived ideas regarding the impact of farmland erosion on the stream system, this overview of the environmental setting of the Saskatchewan River network emphasizes certain key features which must act to limit its importance: the major portion of the water in the main stems originates upstream of the farmed area; there is poor integration of the farmed area with the main stem drainage network; and the most visible sediment source in the basin is clearly bank and valley erosion by streams and rivers deeply incised into the terrain.

2. SOIL EROSION AND SEDIMENT YIELDS

Abundant data exist from WSC (Ashmore, 1986) regarding sediment loads in the Saskatchewan River network, but very little regarding how much of this sediment is from farmland. There are three components to this problem in the context of this review.

The first task is to obtain data for rates of soil erosion on farmland, not in the form of redistribution of soil on farms, but actual loss from the farm.

The second task is to establish how much of the eroded farmland soil is actually delivered to the stream system.

And the third task is to establish how important this off-farm sediment that reaches the stream network is in relation to other sources of sediment.

2.1 Farmland soil erosion: general comments

The topic of soil erosion on Prairie farms has been examined in a general way by numerous investigators over the years. Environment Canada (1985) provided an annotated bibliography dealing with degradation of Prairie agricultural lands. In recent years, Anderson (1975), Geottel et al. (1981), Coote (1983), Agriculture Canada (1983), and Anderson and Knapik (1984) have all provided overviews of the problem, though none of these reports deals with off-farm impacts of the erosion of farmland soils. The last of these reports provided a map of soil erosion risk in the Prairies at a scale of 1:2,500,000, and one of the cost of erosion losses (lost crop value) by Census District (Fig. 2.1).

The Census District map is based on the annual value of crop production lost (1984$) determined as follows: total area (ha) affected by erosion multiplied by yield loss (t/ha) multiplied by crop value ($/t) divided by total area of improved land in the district. The most severely affected districts are those of the Peace River (1: $34/ha; 2: $23.5/ha); the least affected averaged $5/ha. What stands out in the context of the present study is that only two of the ten most affected districts are really in the Saskatchewan River basin: these are the third and fifth ranked districts around the confluence of the North Saskatchewan and Battle rivers. Most of the affected southern districts (6,8 and 9) drain south in the Milk R. system, though there is some drainage north into the South Saskatchewan River.

Most recently, the Land Resource Research Centre of Agriculture Canada, in conjunction with the individual provinces, has produced separate maps of water and wind erosion risk for all three provinces at a scale of 1:1,000,000. The published maps for Alberta were prepared by Coote and Pettapiece (1989) for wind erosion and by Tajek at al. (1985) for water erosion; and for Manitoba by Coote et al. (1989) for wind erosion and by Eilers et

al. (1989) for water erosion. Only preliminary maps are available for Saskatchewan: by Padbury and Coote (1987a,b). These maps are discussed more fully in the separate sections on wind (2.3) and water (2.4) erosion below.

The 1:1,000,000 maps are based on predictive equations for soil erosion, not actual measurement data which are extremely scarce. Both sets of maps are subdivided into polygons which are shaded according to five categories of erosion risk on bare, unprotected mineral soil, ranging from severe to negligible. In addition, each polygon is categorized as to the percentage of the polygon area under cultivation and the degree of protection provided by usual crop management within the cultivated area. Using this additional information, the maps can be interpreted in terms of predicted actual erosion as well as soil erosion "risk" which is the basis of the shading. The equations used to determine risk are discussed in the separate sections dealing with wind and water erosion below.

Snell (1984) has prepared a manual for the regional targeting of farmland soil erosion and sediment loading to streams at a scale of 1:50,000 using plastic overlays for maps of different attributes affecting soil erosion. Sauchyn (1989) has extended the mapping of soil erosion risk to the same scale using geographic information systems software. Initial applications included mapping of water erosion risk in the Rural Municipality of Edenwold, Saskatchewan. Subsequently, it has been used for the application of a runoff-erosion-pollution model (ANSWERS) to simulate erosion and sedimentation in a small (4600 ha) watershed in the same area in response to individual storm events.

Accompanying this progress in the mapping of soil erosion risk, there has been increasing attention to the actual determination of soil erosion rates from farmland in the Prairies. Some of these studies, which treat wind and water erosion separately are discussed in Sections 2.3 and 2.4. Some of the most interesting studies, which do not distinguish between the two processes, are based on Cesium-137 (Cs-137) radioactivity in surface soils; these are discussed next.

2.2 Inferred rates of soil erosion based on Cs-137 data

De Jong and his coworkers at the University of Saskatchewan have relied heavily on the use of Cesium-137 as a means of estimating post-1960 soil erosion rates. This relatively short-lived radioactive isotope (half-life of 30 years) was released into the environment by atmospheric nuclear testing in the 1950's and early 1960's, and was subsequently deposited on land, in the surface soil, through atmospheric precipitation. The majority of fallout in the Prairies occurred in 1960-65 peaking in 1963. The isotope bonds strongly to soil particles and is virtually non-exchangeable. Thus comparison of Cs-137 radioactivity in farmland soils with uneroded native soils should indicate areas of erosion (lower radioactivity per square metre of soil surface)

or deposition (increased radioactivity).

De Jong et al. (1982) reviewed their methodolgy in converting changes in radioactivity to soil losses or gains. Samples are taken of the full Ap (plowed surface soil) horizon (about 15 cm in their areas) with replicate 10cm diameter cores; the samples are air dried and bulked, and the Cs-137 radioactivity determined. The percentage difference in radioactivity compared to non-eroded, non-enhanced soil surfaces is determined (δ). The erosion (deposition) rate E (t/ha/yr) is then computed from the formula:

$$E = \delta \times 1800 \text{ (t/ha)} / 20 \text{ (yr)}$$

where 1800 t/ha is taken as the mass per hectare of an idealized furrow slice of soil 15 cm thick (bulk density of 1.2 t/m^3) and the soil loss (gain) is assumed to have taken place over a period of 20 years. Thus complete loss of Cs-137 radioactivity would be assumed equivalent to a soil erosion rate of 90 t/ha/yr.

De Jong et al. (1983) found little or no Cs-137 redistribution in three uncultivated basins in different parts of Saskatchewan, whereas upper slopes of cultivated basins had depleted radioactivity (soil losses estimated at 10 to 30 t/ha/yr) and lower slopes had increased radioactivity (deposition estimated at 12-40 t/ha/yr). The studies were undertaken in small basins of hummocky topography in which (except for wind erosion) soil redistribution rather than net soil loss would have been expected to prevail.

Martz and de Jong (1985) are reported to have found soil losses averaging 18 t/ha/yr from eroding sites within a fluvially dissected glacial-lacustrine landscape in Saskatchewan, but net soil loss averaged over the area was only 0.6 t/ha/yr. Martz and de Jong (1987) monitored Cs-137 levels at 174 sites in the small Seymour basin, just west of Saskatoon, and found considerable erosion on slope crests and in swales: nearly one-half of the basin had suffered soil loss in excess of 10 t/ha/yr. The mean net erosion of the representative sample of grid sites was, however, close to zero, suggesting that despite marked soil distribution, sediment export was low.

In contrast, Pennock and de Jong (1987), working in the Black Chernozem soil zone about 80 km northwest of Saskatoon, found substantial net export of soil out of the three study areas as well as marked redistribution within them. The three areas were all classified as hummocky terrain (underlain by till, glaciofluvial and glaciolacustrine sediments respectively) and all in the cereal-summerfallow rotation. Representative 7-10 ha tracts of land were sampled for Cesium in the three areas. In each area 83-89% of the grid sites had lost soil with a corresponding minor proportion of sites gaining soil. Average losses at the eroding sites were 27, 16, and 21 t/ha/yr for the three types of deposit. Net soil losses for the three tracts of

land were 17, 10 and 11 t/ha/yr respectively. Comparison of soil losses on individual transects with those predicted by the universal soil loss equation (USLE) indicated that USLE consistently underestimated losses calculated from Cs-137 data by 2 to 9 times. The authors attributed this, in part, to wind erosion.

Kiss et al. (1986) reported high soil erosion rates on almost all cultivated farmland sites (46 slopes) in a study of loamy till soils (Dark Brown soil zone) in undulating terrain near Unity, Sask. The data (Fig. 2.2) are especially significant in indicating very high rates (>20 t/ha/yr) on essentially flat hilltop sites, leading them to the conclusion that almost all erosion was by wind rather than water.

The soil erosion rates indicated by these Cs-137 studies are about an order of magnitude greater than those determined by direct monitoring of erosion by water on experimental plots. The question must therefore be asked whether the higher rates reflect the differences in soil and site setting (in particular the effects of wind) or whether they are artifacts of the computations based on Cs-137 changes.

There are certainly some key assumptions in the Cs-137 procedure which must be addressed. De Jong et al. (1982) discuss some of these. The equation above, for instance, is based on the assumption that radioactivity is uniformly distributed vertically through the Ap horizon. The rates are well above the range 1-10 t/ha/yr noted in the PLUARG studies of farmland erosion in southern Ontario (Coote, 1980).

Mermut et al. (1983) estimated erosion by comparing the thickness of A horizons in a native environment with those on cultivated sites in an area of glaciolacustrine fine-grained soils in southwest Saskatchewan. Their estimates averaged 74 t/ha/yr on a 7.5% slope and 21 t/ha/yr on a 2% slope, and are of the same magnitude as those inferred from Cs-137 data.

2.3 Farmland erosion by wind

The 1:1,000,000 wind erosion risk maps noted in 2.1 above were based on the published soils map in the Soil Landscapes of Canada and on an unpublished model developed from the work of Chepil (1945, 1956) and Chepil and Woodruff ((1963). The model is given as:

$$E = K\ C\ (V^2 - mW^2)^{1.5}$$

where
- E = maximum instantaneous soil movement;
- K = surface roughness and aggregation factor;
- C = factor representing soil resistance to movement by wind;
- V = drag velocity of wind at soil surface;
- m = moisture effect on shear resistance;
- W = wetness or moisture content of the surface soil

The method of determination of the values of the variables at different map sites is outlined in Appendix 1.

The map of Alberta shows high erosion risk in a triangle based on the US border (from the foothills to the Saskatchewan border) and extending north to Strathmore. Pockets of severe erosion risk combined with high percentages of cultivated land are indicated within this triangle especially centred on Lethbridge, as well as outside the triangle south of Brooks and south of Camrose. This pattern in the severity of wind erosion risk appears to be largely controlled by the pattern of wind speed (Fig. 2.3).

In the Saskatchewan part of the basin, pockets of severe wind erosion flank the valley of the South Saskatchewan River (presumably controlled by the pattern of late-glacial sandy deltaic deposits), but the largest area of high and severe wind erosion risk is centred around Lake Diefenbaker, southwest of Elbow. The latter area corresponds with the driest part of the basin with less than 15 mm mean annual runoff.

Overall in the basin, there is a gradual reduction in wind erosion risk from south to north, as would be expected from the generally damper soil conditions in the north.

Referring back to the Cs-137 study by Kiss et al. (1986), it is interesting that the high inferred erosion rates found by those workers are located in an area indicated by the 1:1 million map to be one of largely low wind erosion risk. The exact location of the three areas monitored in the other study by Pennock and de Jong were not reported: their general location appears to have been in areas of low and high (but not severe) wind erosion.

2.4 Farmland erosion by water

The 1:1 million maps of water erosion risk noted in 2.1 above were similarly based on the Soil Landscapes of Canada map, the risk of soil erosion in each polygon being determined by the universal soil loss equation (Wischmeier and Smith, 1965):

$$A = R \times K \times LS \times C \times P$$

where A is the predicted soil loss from water erosion (t/ha/yr), R is the erosivity of rainfall and snowmelt; K is a soil erodibility factor, and LS is a dimensionless slope length and steepness factor. The terms C and P denote dimensionless measures of crop management and conservation practices. In producing these soil <u>risk</u> maps, fields were assumed to be bare (C=1.0) and to have no conservation practices such as terraces or contour strip cropping (P=1.0). Details of determination of the values of different variables at sites through the map area are provided in Appendix 1.

As in the case of the wind erosion risk map, each polygon was rated numerically to indicate the degree of protection actually provided by usual crop management practices. This rating was based on the C factor in the USLE.

To a large extent, the water erosion risk maps reflect topography. The term "severe" was applied to areas with potential soil losses of more than 22 t/ha/yr in Alberta, though the authors emphasize that the indicated soil loss should be treated qualitatively (for ranking purposes only), not quantitatively. The rate representing "severe" in the preliminary Saskatchewan map is not indicated; in Manitoba the rate was taken as 33 t/ha/yr.

In Alberta, severe water erosion risk is found throughout the foothills, and much of the Cypress Hills, as well as on side slopes of the valleys of the Red Deer and parts of the South Saskatchewan River. In Saskatchewan, the Cypress Hills, the Missouri Coteau and the Pasquia Hills all stand out as areas of severe risk, along with valley slopes of the South Saskatchewan River upstream of Elbow, the N. Saskatchewan River, upstream of Prince Albert, Swift Current Creek and Eagle Creek. The location of severe risk areas near to the drainage network might seem to indicate high rates of soil delivery to streams from these sites; on the other hand, it should be recognized that the proportion of the landscape actually cultivated in these valley slope areas is typically only 10-30 percent.

Few direct measurements of water erosion of farmland soils in the Prairies have been made. Van Vliet (Agriculture Canada) is monitoring erosion plots in the Peace District of Alberta and Shaykewich (University of Manitoba) has similar plots in Manitoba. Within the Saskatchewan River basin, the works of Toogood (in the Black Soil zone of Alberta) and Nicholaichuk and Read (in the Swift Current area of Sask) are the most frequently cited. Toogood (1963) found rainfall-induced losses on a summer fallow plot of 10% slope up to 4.5 t/ha/yr. Nicholaichuk and Read (1978) measured average <u>springtime</u> yields (over a period of six years) of about 2 t/ha/yr. No other plot erosion studies are known within the Saskatchewan River basin, except for current work by Anderson and de Jong (Coote, pers. comm.)

2.5 <u>Sediment yields in streams</u>

Hudson and Askin (1987) determined sediment yields and examined sediment sources in the upper Oldman River basin (upstream of Brocket), although only the eastern part of the area is farmed. No other regional studies have been found for other parts of the Saskatchewan River basin. For the basin as a whole, Ashmore (1986) and Ashmore and Day (1988) have summarized suspended sediment data collected by WSC at more than a hundred sampling stations.

These latter authors provide data on sediment yields of sub-basins of the Saskatchewan River which can be compared with farmland erosion rates just discussed. The task must be done with care, however, because the simple t/km²/yr figure determined for a large basin can be misleading. As already noted, large parts of the Saskatchewan R. basin contain hummocky terrain that does not actually drain into the river network, and Mowchenko and Meid (1983) have developed a procedure for determining the "effective" drainage area which is that area contributing flow to the main stream during an average year. Meid and Schoenfeld (1989) provide the latest update on "gross" and "effective" drainage areas for WSC and provincial hydrometric gauging stations in the Prairie Provinces.

Unfortunately, because of the presence of large sediment-trapping reservoirs on many of the Prairie streams, even the "effective" drainage area will, in some cases, overestimate the land area capable of delivering sediment to a sampling station. In these cases, Ashmore determined the land area in a basin that was upstream of lakes and reservoirs and subtracted this from the effective area to determine the "potential sediment contributing area". The assumption was made that the trapping efficiency of the lakes and reservoirs was 100 percent.

Ashmore (1986) separates his discussion of the WSC sediment data according to whether the stations were classed as "full program" (with frequent sampling and interpolation of concentrations between sampling days) or "miscellaneous" (in which sediment loads were predicted from discharge using sediment rating equations). In general, full-program stations tended to have large drainage areas (>1,000 km²) except in the mountains and foothills. The smallest basins with full-program stations in the plains area were on Swift Current Creek with potential sediment contributing areas (less than effective areas) of 858 km² near Leinan and 904 km² near the mouth. Specific sediment yields from these catchments tend to be low, generally less than 100 t/km²/yr (1 t/ha/yr). The yield for Swift Current Creek near the mouth was 74 t/km²/yr.

Perhaps more relevant are the several miscellaneous stations operated by WSC in the plains area because many of these represent smaller catchments and therefore are more directly comparable with farmland erosion estimates. The data for all sites are given in Appendix 2; those for the Plains area are summarized in Table 2.1. Again it can be seen that yields are relatively low (usually less than 50 t/km²/yr or 0.5 t/ha/yr), though there is a large range, amounting to two orders of magnitude.

2.6 Discussion

It is clear that sediment yields in the subbasins of the Saskatchewan River network are significantly lower than most measured and inferred rates of farmland soil erosion in the

catchment.

In part, this is because not all the land area is cultivated, but the proportion that is not farmed is usually extremely small. The 1:1 million erosion risk maps, for example, show that most of the plains area in the two provinces falls into category "5" indicating at least 80% of the land area subject to cropping.

The major explanation must be that most of the sediment that is eroded from farmland is simply not travelling as far as the sampling sites on the river network. In some cases it is clear that soil eroded from the tops of fields is redeposited on the lower slopes of the same fields; soil lost from knolls is redeposited in depressions. The characteristic "hummocky" terrain of much of the plains surface, a reflection of ablation till accumulation during the last glacial episode, is especially conducive to such redistribution of soil rather than its export off farmland. Few data exist, however, to document exactly where this soil is being redeposited. How much simply piles up in roadside ditches and irrigation canals, and is then returned to the land is not known, but could be determined (Sections 4.2, 4.3)

The real question here, however, is not really how much of farmland soil reaches the stream system, but how much of the stream sediment originates from farmland erosion. The disparity between soil erosion rates on farms and sediment yields in streams would, at first sight, suggest that a considerable percentage of streamborne sediment must be from farmland erosion. Ashmore (1986) does not address the issue of sediment sources directly, but Ashmore and Day (1988) comment that in many cases riparian sediment sources (bank erosion) are the dominant factor accounting for the increase in sediment yield downstream in the Plains rivers. Similar sentiments were expressed by Hudson and Askin (1987) in regard to the Oldman River above Brocket.

Assessing the sources of sediment in the drainage network of the Saskatchewan River is no small task. The deeply incised character of many of the streams certainly suggests that bank and valley side erosion is a major sediment source. There is no clear evidence, however, that farmland erosion is unimportant.

A start on the problem could be made by analysing the sediment yield data for small basins listed by Ashmore (1986) and summarized in Appendix 2. The huge range in yields certainly raises questions about the role of land use versus riparian erosion in promoting these above-average sediment yields. It is quite possible that some of these sub-basins have farmland erosion as the major sediment source.

Some indication of the relative contribution of surface soil erosion to the sediment loads of small streams might be provided by the Cesium-137 content of suspended and bed sediment, in

comparison with Cs-137 concentration in upland soils, but no Cs data have been found for stream sediments in the plains area, though Hudson and Askin (1987) did apply the technique in the upper Oldman basin.

On the other hand, small sub-basins do not usually have sediment yields in excess of those in the mainstem rivers. The yield of the Red Deer at Bindloss is about 67 $t/km^2/yr$; that of the South Saskatchewan River above the Red Deer is 90 $t/km^2/yr$; and the North Saskatchewan River at Prince Albert is about 60 $t/km^2/yr$.

This general similarity in yields between the main stems and average yields from tributaries is important, bearing in mind that the bulk of the runoff originates in the headwaters and not from tributaries (Sect. 1.6). With that perspective, most of the sediment load in the main stems would at first sight appear not to originate in tributaries in the farmed Plains area. This may be misleading though: the fraction of the sediment load of a basin provided by the tributaries is given by

$$k \, . \, Y(t) \, / \, Y$$

where Y(t) is the specific yield of the tributaries, Y is the specific yield of the full basin and k is the fraction of the area (not runoff) of the full basin made up by tributaries.

The accurate partition of the sediment load at the downstream limit of a catchment between mainstem contributions and tributary contributions requires data on concentrations (not specific yield) and runoff. The fraction of the total load from tributaries is given by

$$p.Q.c(t) \, / \, Q.c \; = p.c(t)/c$$

where Q is the mean annual runoff at the downstream station, p is the fraction of that runoff coming from tributaries, c is the mean annual concentration (load ÷ runoff) at the downstream station and c(t) is the average concentration for the tributaries.

It has already been noted that p is relatively low (about 0.3). In other words, assuming concentrations in the tributaries are comparable with those in the main stem, the contribution of tributaries to the main stem load is not likely to be more than 30 percent. Assuming that farmland sediment enters the main stem only through tributaries, this places a maximum level on the percentage of sediment from farmland, though how much of the tributary input is from bank erosion, rather than farmland erosion, is still unknown. Unfortunately, data on c(t) for the miscellaneous stations were not computed by Ashmore (1986), only data for the full-program stations. These data for the plains stations are discussed more fully in Section 3.

The issue remains unresolved for the moment, therefore, though some indication of the sediment contribution from the tributaries is provided by examination of mean seasonal loads for which WSC data are available at miscellaneous stations. For example, the station near the mouth of Eagle Creek at Environ (which represents a drainage area of 2,094 km^2 out of the effective area of the N. Saskatchewan basin at Prince Albert of 72,290 km^2) has a seasonal load of only 8 kt (Ashmore, 1986) compared to that of the N. Saskatchewan at Prince Albert of about 3,000 kt. The Battle River at Battleford (9,394 km^2), however, has a seasonal load of more than 1,200 kt though the reliability of this figure was questioned by Ashmore (1986, p. 106). It would seem, from similar comments by Ashmore in relation to other stations in the basin, that, in fact, the accuracy of the sediment load data at some of the WSC stations is insufficient to allow a meaningful assessment of the contribution of tributaries to the sediment load of the main stem rivers. This is compounded by the differing periods of records at the various stations.

Notwithstanding the difficulties of undertaking a regional sediment budget for the basin, there is a general feeling that the increase in sediment yields downstream along the main stems is primarily the result of direct additions of sediment to the main stem itself, rather than from tributary inputs. Even if this view is valid, how much of this sediment input is from bank and gully erosion of valley sides and how much is wind erosion from farmland fields is unknown.

Some indication of just how much sediment can be picked up by the main stem flow from scour of its own channel is indicated by the sediment load of the Saskatchewan River at The Pas. The post-1966 annual suspended sediment load at that station has been estimated at about 2.2 million tonnes (Mt) (Northwest Hydraulic Consultants, 1986), of which only 0.1 Mt come from the upper catchment through Tobin Lake. The rest (apart from about 0.2 Mt from the Carrot River) appears to come entirely from scour of the channel system in the deltaic floodplain area. With a floodplain area of about 8000 km^2, the sediment yield from the Saskatchewan Delta area is about 290 t/km^2/yr, higher than most other subbasins in the Saskatchewan River basin (Carson, 1990).

2.7 Conclusion

Sediment yields from subbasins of the Saskatchewan River, even where they are above-average (up to 200 t/km^2/yr), are an order of magnitude less than soil erosion rates on farmland computed using Cesium-137 data. The implication is that most of the soil eroded from farmland is not transported as far as the stream system. Sedimentation arising from farmland erosion is therefore likely to be more important either on-farm or in those off-farm areas en route to the natural stream system.

How much of the streamborne sediment in the Saskatchewan River basin stems from farmland erosion is a separate issue. The

specific sediment yields of small basins within the Plains area vary through two orders of magnitude (2 to 200 $t/km^2/yr$); it seems probable that some of the highest values are related to farmland erosion, but no proof of this has been established. A closer scrutiny of the existing WSC data would be appropriate.

There is a general feeling that most of the sediment in the mainstem rivers comes from bank scour along the valley sides rather than from tributary inputs, but this is hard to substantiate with the data currently available.

Perhaps the most direct attempt to assess how much of the sediment in the mainstem rivers originates from farmland erosion would be through comparison of Cesium-137 levels in reservoir sediment with those in uneroded farmland soils, as undertaken in the United Kingdom by Peart and Walling (1986) and in the upper Oldman basin by Hudson and Askin (1987). As these authors point out, however, sampling variability of bottom sediments is likely to be great even in a single river reach, because of localized concentrations of wind-blown soil from farmland. Determination of Cs-137 levels in suspended sediment would probably be more reliable, though this would require collection of much larger quantities of sediment than possible with conventional depth-integrated samplers.

3. SEDIMENT CONCENTRATION IN STREAMS

3.1 Background

High levels of sediment in suspension in streams and rivers can pose many different types of problem long before the sediment settles out on the channel bed. In reviewing off-farm impacts in the USA Clark et al. (1985) list the following: turbidity impacts on fish stocks (decreased food supply for visual feeders; inflammation of gills, etc.); damage to hydraulic machinery; decreased aesthetic value of water bodies for recreational use; and increased costs of removal of sediment from water prior to off-stream use.

How important these impacts are in any given area depends on two things: how much use is made of stream and lake water by activities that are sensitive to high turbidity; and how high the concentrations of suspended sediment are. There remains the separate question of how much of this suspended sediment actually originates from farmland erosion. These three points are discussed separately below in the context of the Saskatchewan River basin.

3.2 Use of stream and lake water

A map of important fish and wildlife habitat in the Saskatchewan-Nelson basin (PPWB, 1982) is presented in Fig. 3.1. PPWB (1982, p.114), commenting on water quality for instream use, note that

> "Generally, streams originating in the mountains or PreCambrian Shield are of superior quality to those originating in the Plains, but all of the rivers examined have natural water quality conditions suitable for most uses. Some prairie streams are exceptions, with fish kills occurring in winter due to oxygen depletion under low flow or zero flow conditions."

Significant fish habitat in the Saskatchewan River basin is shown as being primarily in Albertan watercourses draining from the Rocky Mountains, though lower reaches and smaller rivers in Alberta are designated as fish migration routes (e.g. Red Deer R., Battle R., and Vermilion R.). In the Saskatchewan part of the basin, the only important fish areas marked are Lake Diefenbaker and the mouth of Swift Current Creek, the Carrot R. headwaters, streams draining into the North Saskatchewan R. from the Shield area around Prince Albert, and the Cumberland Lake area.

In-stream industrial use of water is primarily for hydro-electric power generation: along the upper Bow R., the upper North Saskatchewan River, Gardiner Dam on the South Saskatchewan River and the Codette and E.B. Campbell dams on the Saskatchewan River. Only the last three would be affected by sediment

generated by agricultural activity in the Plains. However, Herrington (1990, pers. comm.) reports that the Technical Services and Research Branch of Saskatchewan Power regard sediment impacts on turbines and other equipment at hydro-plants as negligible and certainly not measurable.

No adverse comment has been found regarding the impact of sediment on in-stream recreational use of water bodies, though (Section 5) eutrophication of small lakes associated with sediment is a problem in some areas. Herrington (1990, pers. comm.) notes that in some cases, e.g. the Meewasin Valley Authority, Saskatoon, the impacts of sediments on recreation have simply not been documented, and it is therefore difficult to properly assess the situation.

Off-stream uses of water include cooling for thermal-generating plants (Fig. 3.2); municipal consumption (more than 80% is from surface water); industrial use (again more than 80% is surface water) especially chemicals, primary metals, paper, mineral fuels, and food and beverages; as well as farming, primarily for irrigation.

No problems have been noted from sedimentation at any of the thermal-electric plants in the basin (Banga, 1990, pers. comm.), though severe problems have been encountered at Boundary Dam in the Souris River basin due to high silt concentrations in the cooling water (Herrington, 1990, pers. comm.). Smith and Wigham (1990) note that water extracted from Lake Diefenbaker via the Saskatoon Southeast Water Supply System is moved from the upper levels of the reservoir so essentially no sediment enters the canal system. No other information has been obtained regarding the method and costs of filtration of river water at other sites in the basin.

3.3 Sediment concentrations in the Saskatchewan R. basin

Ashmore (1986) has summarized all suspended sediment data gathered by Water Resources Branch of Inland Waters Directorate in the Saskatchewan River basin. The locations of the full-program stations are indicated in Fig. 3.3, and the sediment duration data are graphed in Fig. 3.4 (Oldman and South Saskatchewan River basins) and in Fig. 3.5 (Bow, Red Deer and North Saskatchewan basins).

The only sites which exceed 1,000 mg/L at least 1% of the April-October period are: the South Saskatchewan (at Highway 41, near Lemsford and pre-1967 at Saskatoon), Oldman R. near Lethbridge, Willow Creek near Claresholm (Alberta foothills), Swift Current Creek near the mouth (Fig. 3.4); Red Deer River (at Drumheller and near Bindloss), North Saskatchewan River at Prince Albert and Carrot River near Smoky Burn (Fig. 3.5).

Only three rivers are shown with median seasonal concentrations greater than 100 mg/L: the South Saskatchewan

River (near Lemsford and pre-1967 at Saskatoon), the Red Deer River (near Bindloss) and the Saskatchewan River at The Pas.

The pattern of sediment concentration shown in Figs. 3.4 and 3.5 is relatively simple with levels being generally low in headwater reaches of the main stems and in the streams originating in the Prairies, and high levels in the downstream reaches of the main stems. This downstream increase in sediment concentration is found elsewhere in Canada and seems to result from sediment being added to streams primarily through bank erosion along the river length. The high concentrations in the Red Deer River reflect large supplies of sediment added to the stream from gullies in the badland areas flanking the main stem valley. The high median concentration in the Saskatchewan River at The Pas contrasts with the relatively low 1% exceedance level at the station.

There are, however, exceptions to this general trend: Willow Creek near Claresholm, Swift Current Creek near the mouth and the Carrot River near Smoky Burn drain relatively small drainage areas. The extent to which the high levels in these smaller basins relate to agricultural activity rather than to bank erosion is not known.

Perhaps the most significant feature of the sediment concentration data, however, is the lack of information for small basins. As indicated by Fig. 3.3, the only data available for small basins in the Plains are for Swift Current Creek and the Carrot River. There are no data given for the Battle River, the Vermilion River or Eagle Creek.

There are, however, miscellaneous WRB sediment stations on many of these smaller rivers, for which the data have not been processed to produce concentration-duration curves. A summary of the stations is given in Table 3.1. Some of the maximum concentrations recorded are very high, especially on the Belly River, Meadow Creek, Waterton River, the Battle River and Eagle Creek. Almost 60% of the runoff of the Oldman River at Lethbridge originates in the Belly, St. Mary and Waterton rivers and the high sediment levels at this site (Fig. 3.4) confirm the high levels indicated by the miscellaneous data. It would be instructive to process these data further and to attempt to identify the sources of sediment in these smaller basins.

3.4 Sediment concentrations due to farmland erosion

The high sediment concentrations in the large mainstem rivers are not attributable to farmland erosion. As already noted, most of the flow in these streams originates well upstream of the Plains area. Some runoff and sediment is contributed to the main stems by tributaries, but it seems probable that most of the sediment load in the main stems results primarily from bank erosion by the trunk streams themselves.

The situation in the smaller streams draining only the Plains area is not as clear. Some of these streams have high sediment concentrations as already noted. Whether these high levels are due to farmland erosion or due to downcutting and bank erosion by the streams themselves is unknown. There are very few data available to address this question.

Some data on sediment concentrations in snowmelt runoff from cropped and summerfallowed loam fields in southwest Saskatchewan were included in the report by Nicholaichuk and Read (1978) and are summarized in Table 3.2. Sediment concentrations on fall-fertilized summerfallow fields ranged from 700 to 3,500 mg/L; on ordinary summer-fallow fields they ranged from 1 to 1,900 mg/L and on stubble covered fields the range was 1 to 700 mg/L. Sediment concentrations in return flows in the Bow River Irrigation District reported by Oosterveld and Carefoot (1979) ranged from 20 to 300 mg/L.

It is clear, therefore, that farmland erosion in the Prairies is quite capable of producing high suspended sediment concentrations, though the sampling significance of the data available is limited, and the amount of runoff actually coming from farmland surfaces in these small basins is unknown.

Irrespective of how much of the sediment in small Prairie basins originates from farmland, however, it is not obvious that suspended sediment has a high "impact" on the use of water in such streams. As noted in 3.2, the major use of surface water is on the larger streams where it has already been suggested that most of the sediment originates from riparian, not farmland, sources. Any assessment of impact would therefore require much more information not only on sediment levels and sources, but also on water use in these small basins.

3.5 Conclusions

Sediment concentrations in the main stem rivers can be quite high at times (though small compared to areas in North America with severe soil erosion), but few real problems have been found resulting from these concentrations. In addition, most of the sediment moved in suspension in the large rivers appears to originate from bank erosion rather than losses from farmland.

In the smaller streams, draining only the Prairies, there is not a great deal of data on either sediment concentrations or sediment sources. The limited data available show that sediment concentrations are very variable. Some basins, e.g. the Belly River, have very high sediment levels. How much of this sediment originates from farmland erosion, and how much impact the high sediment levels have on use of stream water are questions which have not yet been answered.

4. SEDIMENTATION OF WATERCOURSES

4.1 Background

Increased sediment production in watersheds does more than simply increase concentrations of sediment suspended in waterbodies. Some of the sediment is too coarse to move in suspension and is transported along the channel bed; other sediment may move in suspension in faster-flowing parts of the drainage network but settle out on the bed further downstream. The increased sediment accumulation on the beds of watercourses can pose many problems.

From a purely physical standpoint, the most obvious effect of sedimentation is to reduce the storage capacity of the waterbody: roadside ditches and canals fill up and need to be cleaned out; streams and rivers may "silt up" more gradually, but eventually lead to increased frequency of overbank flooding and burial of intake pipes. Reservoirs become filled with sediment, reducing the storage capacity available for water, while the deposition of sediment at the head of the reservoir extends upstream producing increased flooding in the valley bottoms leading to the reservoir.

Long before these changes in bed levels become sufficiently extreme as to warrant dredging or other remedial action. more subtle changes take place in the channel bed that can have a dramatic effect on fish. Smothering of gravel with fine sediment reduces the availability of food for many fish species. Clogging of pore spaces between stones blankets fish eggs, reduces the oxygen supply to the eggs through the interchange between streamwater and underflow in the bed, and ultimately asphyxiates eggs and embryos. Not all fish species are affected in this way: the goldeye, one of the most abundant species in the prairies, produces semi-buoyant eggs, and appears to prefer relatively turbid conditions (Donald, 1990, pers. comm.).

In many parts of Canada, the impact of increased sediment production from logging, mining and other human activities on fish habitat has been carefully documented. No reports have been found anywhere in the Saskatchewan River basin, however, regarding similar problems there, arising from agriculture or any other activity. Christensen (1990, pers. comm.), of the Fisheries Branch of Saskatchewan Parks and Renewable Resources, remarks:

"Although we in Fisheries Branch have long been concerned with the effects of sedimentation resulting from the erosion of agricultural land, to date none of our fisheries studies has directly addressed sedimentation issues in the Saskatchewan River basin."

No response has been received from fisheries personnel in Alberta. The issue would seem to be one worth investigating, though the difficulty of separating farmland sediment from other sediment in the channel bottom may not be easy.

In view of the lack of any information on this topic, the remainder of this section is concerned with the direct physical effects of sedimentation, its severity and the degree to which it is related to farmland erosion. Sedimentation is certainly a concern on farms where dugouts, often the main source of farm water supply, may have their capacitiy depleted by sediment accumulation, but the matter is not explored here. The concern of this report is off-farm impacts. Four main categories of waterbody are discussed below: irrigation canals; roadside ditches; streams and rivers; and reservoirs.

4.2 Irrigation canals

As already noted in Section 1, more than 300,000 ha of southern Alberta are irrigated by district irrigation schemes (Fig. 4.1), which involve extraction from mainstem rivers, conveyance along canals and border ditches, flood irrigation of fields and collection of runoff in return flows which drain back to a mainstem river.

Sedimentation in these canals, ditches and drains is certainly an ongoing process from both wind and water erosion of the fields. Oosterveld and McMullin (1979) comment that sediment concentrations in drainwater tend to be highest in the early part of the irrigation season when wind-eroded sediments are flushed out of ditches.

Webber (1990, pers. comm.), General Manager of the Eastern Irrigation District, comments:

> "There are many instances of windblown infill as a nuisance factor to irrigation systems. General O&M costs cover periodic desilting, and some canals are more regularly treated than others."

It appears, however, that records are not broken down in sufficient detail to ascertain the costs specifically associated with sedimentation. The Eastern Irrigation District has only one recent documented event involving removal of wind-blown sediment from its network: about one kilometre of Antelope Creek Canal, Bow City, in the spring of 1987, cleaned out at a cost of about $1,700.

Northwest Hydraulic Consultants Ltd (NHCL, 1982), examined sediment problems associated with the St. Mary main canal, which draws its water from a large reservoir, and is therefore relatively clean initially. Silt and clay are moved through the network but sand is deposited along the canal as inputs accumulate and canal flow diminishes. They note that sediment

clean-out on laterals is conducted on a cycle of approximately five years. "Total length involved may be 300 to 500 km, comprising all sizes of lateral. In larger laterals up to a metre of material may be removed. Five excavation machines are kept busy on this work which is a major operation problem."

Estimation of the total cost of sedimentation in the district irrigation networks might be possible by contacting each district, but it is likely to be a time-consuming, and possibly inconclusive, task.

In any case, not all the sediment that accumulates within the irrigation network is from farmland erosion. Some of it originates in the mainstem river from which the water is extracted. Some of it is from erosion of the ditches and canals themselves as noted in the report by NHCL (1982) on the St. Mary main canal. (In fact, many return flows in Alberta have caused severe degradation and gullying (Spitzer, 1990, pers. comm.)).

The main input to the St. Mary canal was determined to occur between Stafford and Sauder Reservoirs. Estimated historic sediment supply in this reach has averaged about 80,000 tonnes per year. About half of this sediment outflows via laterals and about half deposits in Sauder Reservoir, where 20% of the original live storage has now been lost. Resurvey of canal cross-sections showed, however, that more than 80% of the sediment transported by the main canal originated from bank (and to a lesser extent bed) erosion of the canal, with contributions from land surface runoff and wind erosion of fields being secondary.

Ascertaining the relative contribution of farmland erosion to canal sedimentation is thus likely to be just as difficult and time-consuming as determination of the rates of sediment infill in the first place.

4.3 Roadside ditches

Studies in the USA suggest that the cost of maintaining and repairing road-side ditches, road surfaces and legal drains, as a result of sedimentation can be a major expense in certain areas. Fletcher and Lovejoy (1986) examined data from Indiana and estimated the cost for the state to be of the order of $30 million dollars annually. Their surveys suggested that about 75% of this sediment originated from agricultural land.

No data have been found for the Saskatchewan River basin. Herrington (1990, pers. comm.) reports that Saskatchewan Rural Development does not regard this as a problem. On the other hand, Black (1990) noted that the Manitoban municipalities of Portage la Prairie and Morris spent almost a third of million dollars in 1988 alone on the removal of topsoil from roadside ditches. The problem would seem to be highly variable in both time and space. It would be appropriate to contact the

Engineering Departments of some of the municipalities in the basin, but, as with the problem of canal-filling, it seems unlikely that records are generally kept in sufficient detail to accurately itemize the regional costs involved.

4.4 Stream channel sedimentation

Farmland erosion in the southeast USA in the early 20th century is known to have caused widespread channel infilling resulting in overbank flooding and sedimentation (Trimble, 1975), but there is little, if any, evidence of this in the Saskatchewan River basin.

Herrington (1990, pers. comm.) reports that no case is known to the Water Resources Branch in Saskatchewan of any stage-discharge curve systematically changing over time due to sedimentation, except perhaps the North Saskatchewan River at Prince Albert (Day, 1990, pers. comm.). Some sediment appears to have accumulated at some WRB sites in Alberta, but whether this ongoing aggradation, or simply cyclical fill and scour, is uncertain (Spitzer, 1990, pers. comm.). Little information is available in this context from smaller streams.

Several instances of sedimentation affecting intake structures are documented in the region, but these are not usually the result of widespread aggradation. Smith and Wigham (1990), for example, note that since 1973 the pumpsite for the Chesterfield Flats Irrigation Project on the north side of the South Saskatchewan River, 15 km east of the Alberta border, has been severely affected by sedimentation. The problem, however, results from instability of the channel thalweg in the sandy cross-section (Fig. 4.2). Historically a point of bank erosion (being the outer bank of a gentle meander), the pumpsite bank area became part of a sand bar complex as the thalweg shifted to the opposite bank.

Only one instance of intake structures being affected by widespread aggradation is known in the basin. This is the pumpsite for the Miry Creek Irrigation Project further downstream on the Saskatchewan River. However, the problem was directly related to the impounding of Lake Diefenbaker rather than to increased supply of sediment: the site is located at the head of the reservoir where deltaic accumulation has resulted in the present average bed level now being more than two metres above the invert of the intake pipe (Fig. 4.3).

Both of these examples are unrelated to channel sedimentation from accelerated soil erosion on farmland in the basin. Sedimentation was, of course, widespread in the Saskatchewan River Delta complex downstream of Squaw Rapids prior to the construction of the E.B. Campbell Dam, but since that time the floodplain has become an area of net degradation (Carson, 1990).

4.5 Reservoir sedimentation

Loss of storage capacity in reservoirs through farmland soil erosion is one of the most widely-cited off-farm impacts of agriculture in the USA. In the Prairies, however, the impression conveyed by PFRA staff is that loss of capacity due to sedimentation is not a problem. Unfortunately, however, there is again, little quantitative information to form a reliable opinion, though perhaps more so than in the two previous sections.

Yuzyk (1984) summarized reservoir sedimentation studies in several parts of Canada, including the Prairies. He concluded that "based on the available literature, reservoir sedimentation has not been much of a problem in Canada", except to some extent in parts of British Columbia. Yuzyk (1990, pers. comm.) emphasizes, however, that his study was primarily concerned with large reservoirs, and it could be dangerous to extend the same conclusion to smaller water bodies. The reservoirs examined in the Prairies were:

		Initial capacity (dam^3)	% annual loss
Lake Diefenbaker	1966	9 400 000	0.1
Reid Lake (Duncairn Dam)	1942	103 000	0.04
Avonlea Reservoir	1963	6 035	0.5
Bassano Reservoir	191.	34 300	1.1
Glenmore Reservoir	1932	28 400	0.28
Eastend Reservoir	1936	2 467	0.33
Huff Lake	1939	4 309	-0.08
Newton Lake	1936	11 528	-0.14
Middle Creek Reservoir	1937	16 282	0.09
Altawan Reservoir	1960	6 969	0.18

Only the Bassano Reservoir on the Bow River shows a significant annual loss of capacity, attributed by Yuzyk (1984) to badland gullying. Huff Lake and Newton Lake show negative values (apparent increases in storage volume); these reflect sampling errors in the survey which assume some importance in these two waterbodies because they are located downstream of the Eastend Reservoir (and therefore will have little sediment inflow) on the Frenchman River (Yuzyk, 1990, pers. comm.)

To some extent the % annual loss figures could be misleading, however, because they reflect not only sediment input rates but also initial reservoir volume. The only small reservoirs are Avonlea, Eastend, Huff (downstream of Eastend) and Altawan, though none of these has particularly high sedimentation rates. As noted in the previous section, Sauder (formerly Rattlesnake) Reservoir has lost 20% of its original capacity

(5 400 dam^3) at a rate of 0.7% per year on average, higher than any in the table above except Bassano Reservoir, but little of that was from farmland erosion.

There is some suggestion therefore that sedimentation may be a problem in smaller reservoirs notwithstanding the general comments made earlier. How much of this sediment originates from agriculture is, again, however, a much more difficult question to answer.

4.6 Conclusions

The general impression conveyed above is that sedimentation has not been a real problem along watercourses in the Saskatchewan River basin, and therefore, whatever the contribution made by farmland erosion, it is unlikely to have had a significant off-farm impact in this regard.

On the other hand, few data are available for small watercourses (canals, ditches, streams and small reservoirs) and it is at this scale where sedimentation is likely to be most important and where farmland erosion might be significant. It might be worthwhile attempting to acquire these data by a more intensive investigation, but the task is clearly beyond the scope of the present report.

5. CONTAMINATION OF STREAM SEDIMENTS: NUTRIENTS

5.1 Background

Nutrients are the elements (and their organic derivatives) necessary to stimulate and sustain the growth of flora and fauna. The major macronutrients that limit productivity are nitrogen (N) and phosphorus (P). Both are found naturally in soils, especially near the surface where they accumulate from the breakdown of organic matter, but they are also added to farmland in the form of fertilizers. Runoff from farmland is generally regarded as a significant source of N and P in streams and lakes and may lead to a rapid increase in the plant productivity of water courses.

Eutrophication is the usual term applied to the increase in supply of nutrients to water bodies to a rate at which there is a rapid growth in algae in the water. The nutrients themselves may or may not be undesirable in off-stream use of water: much will depend on its use. For irrigation water, the nutrients may well be beneficial; most industrial uses are not likely to be adversely affected at the concentrations usually found (Hudson, 1971, p.294); whereas if the water is used for domestic supplies there may be problems.

The most important problem, however, is usually associated with algal growth in streams and lakes. Hudson (1971, p. 294) notes that the algae can add undesirable taste or odour to drinking water, can cause problems by clogging intake filters, and can, by their eventual decomposition, affect fish and other aquatic life through depletion of the oxygen content of the water. Munro (1985) reported massive fish kills in some reservoirs of the South Saskatchewan River basin due to sags in the dissolved oxygen levels.

As Gammie et al. (1988) emphasize, the problem with odour is not a trivial one, given that objectionable odour and taste are often equated with health-related concerns and that they can be detected by the human senses at extremely low levels of concentration (parts per billion or even lower). The water supply industry therefore has an obligation to provide water which is not only safe, but which also has low-level odours that are acceptable to consumers (Gammie et al., 1988).

A great deal of attention has been directed to the trophic status of Lake Diefenbaker, the primary source of good quality water in southern Saskatchewan, in which phosphorus has been determined to be the limiting factor to plant productivity in the lake (Weagle and Crosley, 1989). The lake is presently mesotrophic in the upper reaches to oligotrophic (low nutrient level) in the lower reach. Lake concentrations of total

phosphorus (TP) depend primarily on the annual loading of TP from upstream, most of this being in the particulate form (see 5.2).

5.1.1 Nitrogen

The nitrogen compounds are generally far more soluble, and more easily washed off farmland in solution, than phosphorus compounds. Cross et al. (1986), however, determined both dissolved (DN) and particulate (PN) nitrogen in stream samples from the Bow, Oldman and South Saskatchewan Rivers in Alberta and found a definite seasonal peak (Figs. 5.1, 5.2) in PN levels during spring runoff, concurrent with peaks in turbidity (Fig. 5.3) and total phosphorus. At these times PN concentrations (up to 1 mg/L) were comparable with, if not higher, than DN, which peaks during winter and decreases during the growing season through biomass consumption.

Gaskin and Block (1990) found PN to account for more than half of total nitrogen (TN) at high flows on the South Saskatchewan River at Highway 41, and more than 80% of TN during turbid flows from the Red Deer River at Bindloss.

No other data were found on PN in the Saskatchewan basin, only for dissolved nitrogen. For this reason, attention is focussed here on phosphorus, which is usually the limiting nutrient in eutrophication (Coote, 1980).

5.1.2 Phosphorus

Phosphorus tends to be bound to soil grains and is moved off fields into streams primarily through soil erosion. In addition, sediment in streams tends to scavenge any phosphorus that is in solution. Hudson (1971, p. 296) comments that, because of this affinity for phosphorus, sediment is unlikely to act as a storage bank of phosphorus that could subsequently be released into the water. The problem, however, is more complicated than this and is, for example, affected by the oxygen status of the water.

Coote (1980) monitored rates of delivery of total phosphorus being supplied to streams in southern Ontario, and estimated that 60% of the diffuse load of phosphorus entering the Great Lakes came from agricultural land. He found a wide range in yields from 0.2 kg P/ha/yr to 1.5 kg P/ha/yr, the variability being controlled largely by soil and crop type. The lowest yields of both sediment and phosphorus came from cereal fields, pasture and woodland, especially sandy soils, with much higher yields from intensive row crops on clayey soils.

Of the total phosphorus in streams in the PLUARG study, 10-50% was in the dissolved form, and found to be indirectly related to the use of phosphorus in fertilizers and manures. Only about 10% of all phosphorus from agriculture was attributed to livestock. Particulate phosphorus levels in water leaving farmland fields are likely to be higher than background levels in

the soil itself because of preferential entrainment of the fine-grained soil fraction and organic material which contain most of the phosphorus.

5.2 Components of phosphorus in Prairie sediments

Phosphorus in stream water can take many different forms and it is necessary here to define these major groupings (in order to assist in the detection of different sources) and the abbreviations used. At the simplest level total phosphorus (TP) is subdivided into that found in dissolved forms (DP) and that in particulate from (PP). The latter, the focus of interest here, is, in turn, usually subdivided into apatite phosphorus (AP) (which is the orthophosphate incorporated into the crystal lattice of apatite grains from the weathering of sedimentary rock) and non-apatite inorganic phosphorus (NAIP), representing all other inorganic forms including that found in iron and calcium phosphates as well as phosphate ions adsorbed to clay and organic compounds. The other component of PP is organic phosphorus (OP) consisting of compounds of varying complexity which might be derived from decaying stream biota or from surface soils by overland flow.

The NAIP component is sometimes broken down further into exchangeable (adsorbed) phosphates, bicarbonate phosphorus (released by treatment with $NaHCO_3$), and iron- and aluminum-bound phosphates (extracted with NaOH). Some of the phosphorus released by the $NaHCO_3$ and NaOH treatments is organic rather than inorganic. Apatite generally makes up more than 50% of PP in surface farmland soils in Saskatchewan, with another 20-40% being resistant organic complexes (Schoenau et al., 1989). Plant-available phosphorus is largely controlled by the amount of organic and inorganic $NaHCO_3$-P. Where this is less than 60 kg/ha in the top 15 cm, P-fertilization is recommended by the Saskatchewan Advisory Council on Soils.

The issue of P-enrichment in stream water due to removal of fertilizer phosphate from farmland warrants further comment. Losses of <u>dissolved</u> phosphorus are frequently reported as higher from fertilized fields than unfertilized areas, but few data are available for <u>particulate</u> phosphorus. Nicholaichuk and Read (1978) in comparing springtime losses from fertilized and unfertilized fields, commented that losses of sediment bicarbonate phosphorus were not affected by fertilizer.

Certainly, in terms of total PP losses, application of P fertilizer would be expected to have minimal effect. A typical annual addition of 15 kg P/ha in fertilizer might increase the Na-HCO_3-P in the top 15 cm by 50%, but is insignificant in comparison with the total PP in the top 15 cm which is of the order of 1500 kg P/ha.

Moreover, although about threequarters of the cropped land in the Black and Gray Soil Zones (Fig. 1.4) have probably received P-fertilizer, in the less moist (and more erodible) Brown and Dark Brown soil zones, probably a half of the area has never received P-fertilizer (Schoenau, 1990, pers. comm.).

It thus seems unlikely that fluvial sediments are enriched in P because of fertilizer applications. Enrichment in areas unaffected by point sources, if it is found at all, is more likely to result from the slightly higher levels of P in surface soils (compared to exposed subsurface sediment in stream banks) and the selective removal of fines and organic material in surface soil erosion.

5.3 Stream studies: phosphorus in the mainstem rivers

5.3.1 South Saskatchewan River

The mass loading of phosphorus to Lake Diefenbaker was examined by a joint provincial-federal study team (SEPS and IWD, 1988) and summarized by Weagle and Crosley (1989). They determined that the most prevalent form of TP entering Lake Diefenbaker is particulate, accounting for 87% of the input from South Saskatchewan River and 95% of the Red Deer's input. The sediment settles out of the water column within the upper reaches of the lake: PP leaving the reservoir at Gardiner Dam during 1985-88 represented only 3% of PP entering the lake from upstream.

Analysis of bottom sediments from within the reservoir indicated that about 90% of PP was inorganic, equally divided between apatite and non-apatite forms and 10% organic. About half of the bottom sediment was therefore considered to be biologically available under conditions of low oxygen. PP concentrations in the bottom sediment averaged 800 mg/kg (ppm) during the October 1984 sampling, ranging from 669 to 996 mg/kg. Levels were higher in the downriver part of the reservoir than in the area upstream of Riverhurst (Fig. 5.4). This may be related to a fining of the sediment through the reservoir. A survey of the upstream reach in October 1988 showed slightly lower values, in the range of 592 to 771 mg/kg.

Computation of annual loadings of PP between 1975 and 1988 indicated a mean of about 650 tonnes per year from the South Saskatchewan River (at Highway 41) and 480 tonnes from the Red Deer at Bindloss. In-water concentrations of PP were much higher from the Red Deer (as expected from the higher sediment concentrations), but yields (using the potentially available sediment contributing area) were not appreciably different averaging about 0.18 kg/ha.

These in-stream yields are slightly less than those from small cultivated watersheds in Ontario (Coote, 1980), but this in itself does not prove that the PP in Lake Diefenbaker originates

from farmland erosion. On the contrary, there is circumstantial evidence to suggest that most of it is from natural bank erosion and related gullying of valley walls as outlined below. In the case of the Red Deer River, for example, Campbell (1973) monitored natural badland erosion on several experimental plots and, by extrapolation to the full geographical area of badlands along the main stem, concluded that badland erosion, on its own, was more than enough to account for the suspended load actually measured on the Red Deer at Bindloss.

Tones et al. (1980) commented that "on a global basis, the South Saskatchewan River has much lower levels of P and N than most rivers flowing through agricultural regions".

5.3.2 The Bow and Oldman rivers: Alberta Environment study

The study by Cross et al. (1986) of the Bow, Oldman and S. Saskatchewan Rivers upstream of the Red Deer River in Alberta provides a somewhat different perspective on the importance of sediment-associated phosphorus. They found that 50-70% of the total phosphorous in the Oldman River and 80-90% in the Bow River was biologically available, and that most of this (70-80%) was in dissolved form (Fig. 5.5) and virtually all supplied by city sewage treatment plants.

The reduced importance of PP in this study reflects, in part, the emphasis on biologically-available phosphorus; and to some extent it is influenced by the large number of sites on the Bow River. The particulate fraction in the South Saskatchewan River itself, below the Oldman-Bow confluence, was 70-80% of TP (Fig. 5.6). This is still somewhat lower than given by Weagle and Crosley (1989) for the SSR at Highway 41 (87%): but the latter is a 1975-88 mean, while the Albertan data are 1980-81 only. Actual percentages will change from year to year depending on the sediment load: the 1980-81 mean reported by Weagle and Crosley, for example, is only 78%. Both sets of figures may actually underestimate the PP fraction because they are based on interval samples (two, three or four weeks) which may miss short-lived sediment pulses.

Loadings of PP computed by Cross et al. (1986, Table 12) for 1980-81 indicated that two-thirds of the combined Bow-Oldman loading comes from the Oldman River, consistent with the much lower turbidity of the Bow River (Figs. 5.6, 5.7).

5.3.3 The Bow and Oldman rivers: Environment Canada study

Blachford and Ongley (1984) included P in their study of the Bow and Oldman rivers (Figs. 5.8, 5.9, 5.10). They examined the speciation of phosphorus in both bank materials and suspended sediments, and found the bank material composition to be virtually identical to PP speciation in suspended sediments at non-enriched sites during spring time (e.g. Bow River above Calgary: CH in Fig. 5.10). Springtime speciation of PP in

suspension was generally dominated by apatite, except below Calgary (Site 2: Fig. 5.10) where adsorption of phosphate ions below the Calgary sewage treatment plant is assumed to account for the increase in NAI-P. This is consistent with the data of Cross et al. (1986) who also found an increase in PP below Calgary without any corresponding increase in turbidity.

The Blachford-Ongley data show PP concentrations in the suspended sediment and the NAI-component to increase during the summer and fall, though of course loadings are much lower at these times. This seasonal enrichment of sediment with phosphorus (and of the P with both O and NAI forms) as overall sediment concentrations declines includes sites upstream of Calgary on the Bow River (85). Various reasons may be put forward for the change: one obvious one is the probable increase in the clay and organic components of the suspended load at these lower summer flows. The data do not necessarily indicate increased relative importance of farmland sources at this time. This point is pursued in Section 5.5.

One of the interesting components of the Blachford-Ongley study was the use of biomass P loads in the Bow River (estimated by Alberta Environment). They found P concentrations in biomass to average 3,000 mg/kg, comparable with those in the Bow River suspended sediment, and much higher than those in the Oldman sediments. The annual loading moved out of the reach in biomass was calculated to be two orders of magnitude higher than the dissolved and suspended P-loadings. It should be noted that this biomass loading is an inferred quantity based on the assumption that all the aquatic biomass in the Bow downstream of Calgary is uprooted during the high flow period each year and transported downstream to settle out in some sink.

5.3.4 Main stem P sources: conclusion

The conclusion tentatively reached above is that farmland erosion is not a major source of phosphorus in the main stems of the Saskatchewan River network: the Oldman, Bow, Red Deer, South Saskatchewan and North Saskatchewan rivers. Eutrophication is rarely a problem there at the present time, and where it does occur it is more related to dissolved phosphorus emitted from sewage plants in the major urban areas. The same conclusions do not necessarily apply to smaller streams in the basin, however, whose runoff is not dominated by alpine sources. These are discussed next.

5.4 Phosphorus in tributary watercourses

The streams and lakes of the Prairies that are tributary to the mainstem rivers, and which derive their water entirely from Prairie sources, are generally regarded as much more eutrophic than the larger rivers, though the extent of this due to agriculture has not been closely documented.

5.4.1 Saskatchewan

SEPS and IWD (1988) express the view that eutrophication is the most significant water-quality-related problem in the South Saskatchewan R. basin with many water bodies now eutrophic from both natural and man-made causes.

Johnson (1978), discussing land use and the fish resource in Saskatchewan, commented:

> "Nutrients from domestic, industrial and agricultural wastes combined to produce marked eutrophication of lakes and streams (in the Qu'Appelle system). This eutrophication has been accompanied by aggravation of anoxic (low oxygen) conditions resulting in fish kills; limited production of certain species and changes in fish species favouring "rough" fish, even though total fish biomass has increased; toxic algal blooms probably causing fish deaths; (and) possibly an increase in fish diseases and tainting of game fish flesh."

Miles and Sawchyn (1988), in describing the fishery survey of the South Saskatchewan River and its tributaries, take a somewhat less negative position in this region, pointing out that small eutrophic reservoirs provide ideal conditions for the production of yellow perch, walleye and northern pike, and also comment that the main stem downstream of Saskatoon "thrives" from the effect of nutrient enrichment from the city.

Johnson (1978) cites Hammer (1972, 1973) to the effect that soil erosion, an unnecessarily high rate of fertilizer application and irrigation return water all add nonpoint nutrients which are a major factor in the "rush eutrophication" of Saskatchewan lakes, emphasizing the importance of soil erosion (Johnson, 1972).

Liaw (1979, 1982) undertook a comparison of trophic status in close to sixty Saskatchewan lakes, classified according to land use in the contributing drainage area: (1) forest, (2) mixed woodland and agriculture, and (3) agricultural-urban. Though only a few of these lakes were in the Saskatchewan River drainage basin (Wishtikan, Atton, Jackfish, Murray and Perch in the Battleford area; Blue Heron, Von Mehren, and Round near Prince Albert; Pike and Blackstrap south of Saskatoon), the results are important in representing one of the few regional quantitative studies of land use and trophic status. Summertime TP levels in woodland lakes were low and averaged 0.016 mg/L; those in mixed wooded-cultivated averaged 0.036 mg/L; and those in agricultural-urban watersheds averaged 0.30 mg/L. Included in the second group were Blackstrap (12 km^2) and Pike Lake (2.6 km^2). No breakdown between PP and DP was given. It is assumed that most TP in the lakes was dissolved, but ultimately in large part originating from sediment.

The increased trophic status associated with agriculture is clear. The major increase is in the third group, however, which includes lakes in the Qu'Appelle valley and elsewhere in which the effects of cottage development and municipal sources are hard to separate from agriculture. The TP concentrations in this group are comparable with summer levels in the Bow River.

5.4.2 Alberta

Similar opinions have been expressed regarding small lakes in the Alberta part of the basin, particularly in the Battle River valley, Driedmeat Lake and Coal Lake (Bjonback, 1990, pers. comm.).

Alberta Environment has examined farm-related eutrophication in two lakes west of Edmonton: Lake Wabamun and Lac La Nonne.

Lake Wabamun is a large (about 75 km^2) recreational waterbody surrounded mostly by mixed agricultural-wooded land, with smaller watersheds covered almost entirely by forest. A phosphorus budget during 1981-82 (Mitchell, 1985) indicated the major source of TP to be runoff from surrounding watersheds, with yields from the mixed agricultural-wooded basins (0.19 kg/ha/yr) being double those of the forested watersheds. Average annual TP concentrations in streams draining the farmed watersheds ranged from 0.2 to 1.4 mg/L, compared to 0.1 to 0.4 mg/L in wooded basins. The second major P source was release of phosphorus to the water column from sediments on the bed of the lake (Fig. 5.11), especially during the period of low oxygen levels in the lake during late summer. Again the impact of soil erosion is evident. Sewage from cottages had a minimal effect on P levels.

Much of the Lake Wabamun area farming appears to be livestock, however, and studies in watersheds leading to Lac La Nonne (Mitchell and Hamilton, 1982) have shown TP yields from these basins to be proportional to the density of livestock grazing, the yields ranging from 0.1 to 1.4 kg/ha/yr, substantially higher than in the Lake Wabamun basin.

How representative the TP yields and concentrations in these two areas are compared to cereal cropping further south is unclear, but the studies are extremely significant in unequivocally emphasizing the major role of farming in the eutrophication of lakes and small streams, as well as the long-term contribution provided by bottom sediments.

No information has been obtained regarding the trophic status of small streams, lakes and reservoirs in the Oldman-Bow basin or the Red Deer basin of the Alberta Prairies. The study by Blachford and Ongley (1984) was primarily orientated to the main stems, but did include limited data for tributaries of the Oldman during early summer conditions (Figs. 5.9, 5.10).

These tributary data show higher mg/L PP concentrations on the Belly R. (0.11 mg/L) than on the main stem, while mg/L PP concentrations on the St. Mary R. are smaller (0.03). The differences are simply a reflection of the higher total suspended sediment concentrations on the Belly than on the St Mary. On the other hand, the St. Mary sediment is much richer in phosphorus (almost 6000 mg/kg) and is virtually all organic and non-apatite inorganic, whereas the Belly sediment contains only about 10% as much P, and most of this is unavailable biologically. The contrast in sources of sediment and phosphorus between the two catchments is the kind of information which (while beyond the scope of the Environment Canada study) would have been useful in the present review of the effect of land use on stream sediments.

5.4.3 Manitoba

Gregor and Chako (1987) included the Saskatchewan River above Carrot River in a 1970-79 study of 24 sites along the borders of Manitoba. The data show mean annual concentrations of TP and TN there to be substantially lower than at sites on smaller rivers to the south: 0.06 mg/L TP compared to >0.2 mg/L for the Souris, Qu'Appelle, Red and Pembina rivers; and 0.79 mg/L TN compared to >1.5 mg/L on the Antler, Assiniboine and Souris rivers. Settling of sediment in Lake Diefenbaker and Tobin Reservoir would be expected to lower TN and TP concentrations on the lower Saskatchewan River.

5.5 Phosphorus losses from farmland soils

Not a great deal of information appears to exist on P losses from farmland in the Prairies. Nicholaichuk and Read (1978) examined the loss of nutrients from farmland plots in the Swift Current area of Saskatchewan during snowmelt, but did not determine the particulate component.

Tones (1987) reviewed export coefficients (average annual yields of total phosphorus in kg/ha/yr) for different types of land use as they might apply to the Lake Diefenbaker watershed: these ranged 0.2-0.5 kg/ha/yr.

In the PLUARG study in Ontario, PP concentrations were determined directly in runoff samples from fields, averaging 1660 mg/kg (Miller et al. 1982), much higher than concentrations in surface soils (which averaged 730 mg/kg) and in stream bank sediments (which were generally less than 500 mg/kg) (Knap, 1978). This P enrichment of farmland runoff appears to be due to the preferential entrainment of P-rich organic and fine grained sediment in the overland flow. It makes the task of quantifying the sources of stream sediment phosphorus very difficult based on mg/kg P concentrations in the source areas without supplementary data on grain size and the organic component.

The only data for P losses from farmland in the Prairies that are based on direct measurement are for irrigation return flows. Large areas of the Oldman and Bow basins are subjected to irrigation during the summer (Fig. 5.12). Oosterveld et al. (1978), for example, reported that return flows made up about 20% of the South Saskatchewan River flow at Medicine Hat during the summer months. These return flow data may be useful in two contexts. Firstly they provide insight into the direct importance of return flows on summer P loadings in the receiving waters of the main stems. And secondly, although springtime yields (kg(P)/ha) and in-water concentrations (mg/L) in farmland runoff would be expected to be much higher, the return flow in-sediment concentrations (mg/kg) should prove useful in comparing farmland soils with stream sediments.

Oosterveld and Carefoot (1979) examined nutrient and sediment concentrations in return flows from the Vauxhall and Hays irrigation blocks of the Bow River Irrigation District. They found average TP concentrations up to six times the Alberta water quality objective of 0.05 mg/L, most of this being in particulate form (Table 5.1). These figures are comparable with summer-autumn concentrations on the Bow and Oldman Rivers.

The sediment and PP in these irrigation return flows was attributed to erosion from the banks and bottoms of the drains rather than farmland erosion. A close inspection of the data suggests that much of this in-drain sediment, however, may have originated from the fields. In-sediment P concentrations (calculated from Table 5.1) were noticeably higher in the drains (813-1562 mg/kg) than in the irrigation canal (374 mg/kg) based on the Ronalane wasteway sample.

Data for in-sediment P concentrations from surface croplands in Saskatchewan (Table 5.3) show values in the range 350 to 700 mg/kg, and slightly lower for subsurface fractions (Roberts et al., 1985; Schoenau et al., 1989). These are comparable with the values in the irrigation canal, but much less than those found in the return flows, suggesting that the latter have been enriched by preferential removal of P-rich sediment from farmland soil.

A more detailed study of a 3000 ha flood-irrigated basin (Oosterveld and McMullin, 1979) showed TP concentrations in field runoff (0.18 mg/L) well above those in drains (Table 5.2) where settling out had occurred. Annual yields of TP were 0.2 kg/ha. No P-fertilizer had been applied to these fields in either year. The ortho-P (dissolved inorganic phosphate) is seen to make up 45-65% of the TP in the field runoff. A detailed phosphorus speciation was not given.

These sparse data do suggest that sediment losses from Prairie fields are enriched in P compared to background sediments. Surface soil erosion would therefore be expected to produce increased P-levels in small watercourses.

In relation to mainstem rivers, two key questions need addressing. How important is this return flow sediment in terms of P concentrations in stream sediments during the summer? And what do the data indicate in relation to sources of P in stream sediments during spring, when PP loadings are greatest?

In terms of the first question, Cross et al. (1986) concluded that irrigation produced a net reduction in the P-loading imposed on the Bow River in 1980 and 1981. The return flow volumes were lower than the volumes of water abstracted from the river, and in some cases the P concentrations were also lower in the return flows. The latter point results from the the high levels of DP in the abstracted water associated with the location of the diversion works downstream of the Calgary sewage plant.

Joseph and Ongley (1986) also concluded that irrigation return flows do not lead to P-enrichment of the Bow and Oldman rivers during summer. They noted that annual mean TP of return flows (830 mg/kg) is less than summer and autumn TP river concentrations by a factor of 2.5. In particular, while A-P was relatively constant between seasons in the Bow and Oldman, summer and autumn river samples were enriched, relative to return flows, by factors of 5x for NAI-P and 4x for O-P. Again, the authors attributed this enrichment to the increased dominance of PP from sewage effluent from Calgary and Lethbridge as natural sediment concentrations fell in the rivers.

In relation to the second question, the Joseph-Ongley report noted that annual mean concentration and speciation of PP in irrigation return flows were very similar to those in spring time suspended sediments in the rivers (Fig. 5.13: compare with Fig. 5.10): A-P averaged 457 mg/kg, NAI-P 167 mg/kg and O-P 264 mg/kg in the return flows, totalling 888 mg/kg. They concluded:

> "Because natural erosion during spring contributes large quantities of sediment to these rivers, we conclude that suspended sediments in irrigation return flows throughout the 1980 sampling period are not, as a group, enriched relative to naturally occurring suspended sediments in these river systems." (Joseph and Ongley, 1986, p. 653).

In short, PP values in return flows were comparable with those in springtime mainstem flows as well as in Lake Diefenbaker bottom sediments. All, however, are above the levels just noted for surface soils in the area, and some caution is required in making these comparisons

The higher levels in the Lake Diefenbaker sediments may be the result of several processes, including the preferential deposition of coarser sediment (sand) in the upstream part of the reservoir. As the data of Tiessen and Stewart (1983) indicate, the silt-clay fraction of the soil is enriched (in both inorganic and organic P) relative to the bulk sediment (Table 5.4).

The same explanation may hold for the mainstem suspended sediment samples which, taken with a pump-centrifuge sampler at shallow depth in the water column, would have underestimated the sand component in the flow. It is possible, therefore, that PP levels in return flows (and other surface runoff) are in fact enriched relative to springtime mainstem flows, the latter sediment (when not biased to near-surface water) being diluted by scour of somewhat coarser bank sediment.

5.6 Conclusions

The present evidence suggests radically different off-farm impacts in terms of nutrient loading according to whether the receiving waters are main stem or tributary.

The main stems, in which most of the runoff originates in the headwaters beyond the agricultural zone, appear to have sediment loads derived largely from natural erosion of "subsurface" sediments, i.e. undercut channel banks and valley slopes. Nutrient-rich sediment may enter the main stems from tributaries (e.g. the St. Mary River) and irrigation return flows, but the impact of these inflows appears to be small relative to the "dilution" effect of background sediment from natural sources. This appears to be especially so during spring runoff. In summer, main stems appear to be enriched relative to return flows, probably because of the large amount of sewage waste from urban areas in comparison with natural sediment in suspension.

Visual observation indicates that much sediment from farmland, found in drains and ditches by the end of winter, is indeed flushed into the main stems during spring. The magnitude of this input appears to be insignificant, however, in comparison with natural erosion of subsurface sediment, especially through stream bank erosion.

It would be logical to compare mg/kg PP concentrations of reservoir sediments with those for different possible sources (channel biomass, channel banks, valley slopes, farmland soils) to assess the degree to which main stem sediments are enriched (relative to natural sediments) or impoverished (relative to farmland soils). The latter comparison would be analogous with the Cs-137 comparison previously suggested in connection with sediment loadings. The P concentrations in farmland soils appear to be only slightly higher than those in subsurface sediments, however, in contrast to Cs-137 levels, and therefore the approach is unlikely to offer useful data.

In terms of tributary watercourses, there is a general acknowledgment that these waters tend to be more eutrophic than those of the main stem. There is also circumstantial evidence in many cases linking the increased trophic status to agriculture, and in several cases direct loading budgets have been done to

prove the point. Eutrophication of small lakes in this part of the Prairies is accelerated by their shallow depth and hence high summer temperatures. The problem is not a minor one given the dependence of many rural communities on such lakes for their water supply (Gummer, 1990, pers. comm.) and given the scarcity of recreational waterbodies in the area.

6. CONTAMINATION OF STREAM SEDIMENTS: HEAVY METALS

6.1 Background

Heavy metals are found naturally in soils, but may also enter the land through farming activity, especially as part of fertilizer applications and from pesticides (in which metals act as the host for a series of anionic chlorohydrocarbons). Runoff from farmland soils may therefore be expected to be enriched in metals, especially in particulate form given the tendency of many metals to bond to soil particles. This could, in some cases, be a matter of concern, given the toxicity of many heavy metals to organisms.

As part of the PLUARG study in Ontario, Ihnat (1978) examined levels of copper, zinc, cadmium and lead in streams draining small agricultural areas, and concluded, from the similarity of the data with those from non-farmed areas reported in the literature, that there were "no gross effects of agricultural practices on trace element levels in streams" in the area. Mean concentrations in the suspended sediments at different sites ranged 25-112 mg/kg for copper, 50-290 mg/kg for zinc, 0.1-6.2 mg/kg for cadmium and 21-280 mg/kg for lead. Not all of this metal is bioavailable.

Coote (1980) summarized findings of the PLUARG study and reported that fertilizer contributions of lead, mercury, arsenic, cadmium and selenium were all negligible.

6.2 Saskatchewan River basin data

Studies of heavy metals in the surface waters of the Saskatchewan River basin have been included in several recent reports, and are summarized below.

6.2.1 Bow and Oldman rivers

The results of studies of metal pathways in the Bow and Oldman Rivers were included in the report by Blachford and Ongley (1984). The metals analyzed were Mn, Ni, Fe, Cu, Pb, Zn, Co, Cr, Cd and Hg. They noted that most filterable and many "total" measurements as µg/L in water were below conventional detection limits, except in spring when concentrations increased about ten-fold because of higher concentrations of suspended matter. Estimates of total seasonal loads of metals (1980) for the Bow River and Oldman River just upstream of their confluence are provided in Table 6.1 and emphasize the large proportion of the total load that is associated with sediment. The table also indicates the large quantities in biomass storage in the reach too.

During spring, whole-water (unfiltered) metal concentrations tended to increase downstream, in concert with the increase in sediment concentration along the stream network. In all three cases (Zn, Pb and Hg), however, springtime metal concentrations per gram of stream sediment (Fig. 6.1) were only slightly greater than those determined on sediment samples from undercut banks along the stream course (Table 6.2); though Pb was distinctly higher in the Bow. The implication seems to be that almost all sediment and associated metals in suspension during spring high flows comes from undercut channel banks rather than farmland erosion.

Metal concentrations in the stream sediment increase during summer and often remain high during autumn. This was attributed primarily to the much lower total sediment concentrations during these months allowing particulate metals from point sources in Calgary and Lethbridge, including urban storm runoff, to dominate.

The Blachford-Ongley data do not, therefore, indicate a significant supply of metals to streams from farmland areas, or, more correctly the contribution from those sources in the two main stems does not substantially exceed what would be expected from background sediment sources. The key question, however, is in relation to bioavailable, rather than total, metal inputs to the stream system. The authors cite the work of Joseph (1982) to indicate that most sediment-associated loads from irrigation-return flow sediments seem to be largely unavailable for biological uptake (Section 6.3).

6.2.2 South Saskatchewan River Basin

Tones (1988) has reviewed water quality data in the Saskatchewan part of the South Saskatchewan River basin, using the ESQUADAT and NAQUADAT water quality databases and relevant publications of the Prairie Provinces Water Board and other agencies. She reached the following conclusions regarding levels of metals in relation to CCREM (1987) water quality guidelines:

i. **Arsenic, barium, beryllium, cobalt, mercury, molybdenum and vanadium:**

Concentrations were below the CCREM guidelines for the most sensitive use. Mercury contamination of fish and sediments is known to exist, however, notwithstanding the levels noted in the water quality databases.

ii. **Cadmium, nickel, selenium and silver.**

Concentrations were slightly above the detection limit, with a small number of samples from a few locations exceeding the CCREM guidelines for the most sensitive use. The risk to aquatic biota (the most sensitive use) was, however, considered minimal.

iii. **Aluminum, lead and zinc.**

Concentrations exceeded the CCREM guidelines to protect aquatic life (the most sensitive use) in many of the samples. These were associated with clay size suspended solids in the water. The concentrations were possibly marginally toxic to aquatic biota.

iv. **Chromium and copper.**

Concentrations exceeded the CCREM guidelines to protect aquatic life in more than half the samples. Neither element was regarded as toxic to fish directly, but the toxicity to inverterbrates might affect their food supply. Maximum concentrations of chromium approach the guideline for drinking water quality.

v. **Iron and manganese.**

Concentrations exceeded the CCREM guidelines for drinking water supply. Again, the high concentrations were associated with clay size suspended sediment. Aesthetics (spotting of laundry etc.), rather than toxicity, is the main problem in this case.

The role of soil erosion, in general, and farmland runoff in particular, in contributing to the presence of these metals in stream water varies. Lead is thought to originate primarily from municipal and industrial runoff and wastewater. Chromium and copper will also be contributed by industry, though both are found naturally in soils. Soil erosion is likely to be responsible for much of the aluminum, iron, manganese and zinc, but natural erosion (gullying and bank erosion) is just as likely a source as farmland erosion.

How much of the metals are actually bioavailable is another issue, and one for which only limited data exist. Overall, it does not appear that heavy metal contamination from farmland erosion is an important issue in this region. The data are biased, however, to the mainstem South Saskatchewan River, for which most of the sediment is probably from bank erosion, both along the reach and upstream and in tributaries. Few data were available for the Swift Current Creek subbasin.

6.2.3 The North Saskatchewan River

Ongley et al. (1985) included heavy metals in their study of low-flow (1983) and 1984 spring conditions along the North Saskatchewan River and in Tobin Lake. The data are summarized graphically in Figs. 6.2-6.4 for copper, zinc and mercury.

All three elements show elevated concentrations in water during spring conditions, associated with the much higher levels of suspended sediment at that time, though this is less pronounced for mercury. Concentrations of the metals in suspended sediment increase abruptly downstream of Edmonton (Site 2) during low-flow conditions. All three also reach peak

concentrations in the water downstream of Edmonton during spring conditions, again suggesting increased contamination of suspended sediment from urban-industrial sources. On the other hand, concentrations of the elements in the sediment are remarkably stable along the reach during high-flow conditions, with no significant increase below Edmonton. The implication appears to be that, during spring runoff, the metals are found in ambient sediment which itself abruptly increases in concentration below Edmonton.

Springtime metal concentrations in the sediment are about 40 mg/kg (Cu), 100 mg/kg (Zn) and 0.6 mg/kg (Hg). The first two are slightly higher than springtime levels in suspended sediment in the Bow-Oldman Rivers noted previously (Fig. 6.1), and slightly higher than metal concentrations in bank sediment in the latter area (Table 6.2). Comparison with data for heavy metals in surface soils in the Prairies (Table 6.3) again illustrates that, while Zn is at background levels, and Cu is only slightly above background, Hg levels in sediment in the North Saskatchewan River are an order of magnitude higher than in farmland soils. This is consistent with the role of sediment as a scavenger of Hg dissolved in water, but the source of the mercury remains uncertain.

The role of farmland sediment in the contamination of fish by mercury is not clear. It has been suggested that the former use of mercury as a fungicide in the 1960's may still constitute a widespread source. AEC (1983a) reported mercury concentrations in bottom sediments (sieved into two fractions: 75-250 µm and < 75 µm) for the mainstem North Saskatchewan River in Alberta. Levels were about 100% higher downstream of Edmonton compared to upstream (Table 6.4). This higher level persisted throughout the river to at least the Saskatchewan border, the river draining through an area that is primarily agricultural. These levels in the bottom sediments (about 0.1 mg/kg) are comparable with those found by Ongley et al. (1985) for bed sediment in the Alberta reach downstream of Edmonton; they are 2x-3x higher than in bank sediments and farmland soils, but are almost an order of magnitude less than in suspended sediment.

Concentrations of mercury in whole-water samples (AEC, 1983a) varied appreciably along the North Saskatchewan River, and over time, ranging up to 1.22 µg/L. Much of the variability is presumably due to changes in sediment levels (mg/L) as well as proximity to point sources. No data were available on fluctuations in overall sediment concentrations. AEC noted that mercury levels seemed to be highest near industrial discharge sites.

The most heavily contaminated fish are migratory and it was therefore thought likely by AEC that uptake of mercury occurs primarily in Saskatchewan from industrial sources (pulp and paper mills, chlor-alkali plants, etc.) than from urban-industrial waste or farmland runoff in Alberta. Unfortunately no AEC data

are available for Hg concentrations in bottom sediments in the Saskatchewan part of the river. The trend for both copper and zinc (Fig. 6.3-6.4) is, however, for decreased concentrations in the bottom sediments downstream of the Alberta border; this is consistent with the Hg levels reported by Ongley et al. (1985), all being less than the detection limit.

Harrison and Waite (1988) report Hg concentrations in water, sediment and biota in the South Saskatchewan River and downstream. Upstream of Saskatoon, levels were 0.01 to 0.06 mg/kg, with higher levels downstream of the chlor-alkali plant in Saskatoon, although concentrations have decreased in the last decade. In fact, Hg concentrations in the bottom sediments downstream of Saskatoon decreased from 0.95-2.45 mg/kg in 1971 to < 0.03 mg/kg today, and are now at background soil levels. Sediments in Tobin Lake have Hg levels of 0.08-0.24 mg/kg, and levels in Cumberland Lake ranged 0.01 to 0.08 mg/kg. Tobin Lake thus appears to be a sink for excess sediment-associated mercury from the North Saskatchewan River.

6.3 Metals in Tobin Lake sediments

Warwick (1985) examined the incidence and severity of antennal deformations in chironomidae larvae (benthic organisms that are known to graze on contaminant-laden organic detritus) in Tobin Lake. He noted these deformations to be six times more frequent than those found in Last Mountain Lake and believed this to be due to the much higher pollutant loading into Tobin Lake.

Warwick and Tisdale (1988) have examined the morphological characteristics of chironomidae larvae in two different parts of Tobin Lake, one site downstream of the Saskatchewan River entrance and one site in a backwater area that receives drainage from the much smaller Petaigan River. The frequency of morphlogical deformities was 32% at the latter site but 41% at the former. Bottom sediment data (reported by Warwick, 1990, pers. comm. and given in Table 6.5) show metal concentrations to be either the same at the two sites or lower for the Saskatchewan River entrance: the exception was mercury which averaged 0.084 µg/g near the mainstem entry but only 0.046 µg/g in the side area zone. The source of the increased deformities in the chironomidae larve in sediments from the mainstem remains unknown.

It may be significant, however, that metal levels (except for Hg) tend to be higher in the side bay location of Tobin Lake, the levels for Ni, Cu and Cd being two to three times higher than in the sediments at the Saskatchewan River entrance (Table 6.5). On the assumption that the sediment in the side bay location is largely from the agricultural watershed of the Petaigan (rather than from dispersal of the suspended load of the Saskatchewan River in the lake), these higher metal levels may be more representative of sediment lost from Prairie farmland. The lower levels in the mainstem area presumably reflect the "dilution" by natural sediment (particularly the sand fraction),

notwithstanding the point source loading from industrial and municipal sources upstream. The reason for the very high levels of chromium at both sites is unknown.

6.4 Sediment quality in irrigation return flows

Joseph and Ongley (1986) examined metal concentrations in sediment carried by irrigation return flows leading into the Bow and Oldman Rivers. The seasonal fluctuations for different metals (no Hg data) in the various return flows are shown in Fig. 6.5.

With the exception of Cu, Mn and Fe, concentrations in the water of the return flow were below detection level, so that the percentage of total metal in particulate form could not be accurately determined. In the case of Mn and Fe, at sediment concentrations greater than 100 mg/L, essentially all (>99%) of the metal in the water was in particulate form, the proportion becoming less at lower concentrations. A similar trend was noted for Cu, though the maximum amount in particulate form was only 67 percent.

The levels in Fig. 6.5 should be compared with those for the Bow-Oldman river suspended sediments (Fig. 6.1) and those in bank materials along the same rivers (Table 6.2) and in surface soils (Table 6.3). Though levels in the return flows are comparable with those in soils, it is evident that in almost all cases, the metal concentrations in the return flow sediment are no higher than those in springtime suspended sediment or in channel bank sites.

The speciation of the metals in the return flow also supports the view that they originate from natural sources. This is indicated in Fig. 6.6. The labile ($MgCl_2$ exchangeable) component was extremely small. The fraction associated with carbonates (extracted with 2.5% HCl) is much bigger, but the main component is generally that bound up in silicate structures and only released after treatment with HNO_3-HCl.

Thus, irrespective of the proportion of the sediment load which originates from farmland erosion, the metal concentrations in farmland soils and channel-bank sediments tend to be so similar that little, if any, "enrichment" of metals in suspended sediment would be expected from farmland erosion.

Joseph and Ongley (1986) conclude that all the evidence suggests that "suspended sediment-associated metals from irrigation return flows are not anthropogenically enriched relative to naturally occurring sediment chemistry. It is unlikely, therefore, that suspended sediments in these return flows exert a deleterious effect on riverine metals and phosphorus chemistry in the South Saskatchewan River." The statement could probably be applied to all runoff from farmland in the basin.

6.4 Conclusions

The evidence presently available provides little support for the view that farmland runoff has increased metal concentrations in the sediment of streams in the basin. The data are, however, biased to the mainstem rivers. The sediment in these rivers is probably largely from bank erosion and gullying so that any metal contribution from farmland would be masked by the background levels in non-farmland sediment.

Nonetheless the same conclusion probably applies to smaller basins too. Irrigation return flows (which should be typical of farmland runoff in general) do not show metal levels in suspended sediment any higher than those found in the mainstem rivers of the basin, so that smaller streams (in which farmland runoff and sediment may be relatively more abundant) should not show elevated metal levels either. This finding should perhaps have been expected given the similarity of metal concentrations in farmland soils and natural geologic sediments found in the basin (TAble 6.3). The statistical representativeness of the data in Table 6.3 is, however, unknown. Little is known also about the relative bioavailability of metals in return flow sediments compared to stream sediments.

It may be significant that metal concentrations in the bottom sediments of Tobin Lake are up to three times higher in the side-bay sediments, assumed to be derived from the small Petaigan River basin, than in the sediments at the entrance of the Saskatchewan River. Additional monitoring of small basins does seem to be warranted.

Mercury appears to be widespread in the sediments of the mainstem, and to varying degrees in fish, but the source of the mercury has not been clearly defined. It seems more probable that the main source is industrial, rather than farmland soil, but the matter remains unresolved.

7. CONTAMINATION OF STREAM SEDIMENTS: ORGANIC COMPOUNDS

7.1 Background

The major deleterious organic compounds in soils are pesticides (herbicides, fungicides and insecticides) applied deliberately to the land, and atmospheric fallout of compounds of industrial origin such as polychlorinated biphenyls (PCBs).

The application of pesticides to soils and plants poses problems because of the possibility of dramatic accumulation of the chemicals at higher levels in the food chain. Mellanby (1967), for example, cited an American lake where concentration of DDDs (a relative of DDT) increased from less than 0.015 ppm in the water to 5 ppm in plankton, 10 ppm in small fish, up to 100 ppm in larger fish and over 1500 ppm in predatory fish-eating birds, this level being fatal.

Admittedly, the increased environmental awareness of government agencies during the 1970s led to restrictions in the use of many of these pesticides, and to the complete banning of some such as DDT, but many of these pesticides persist in the soil long after their application. Nash and Woolson (1967) documented in Maryland, for example, that, 20 years after application, up to 40% of some pesticides were present in the soil.

Pesticides are important in the present study because, in many cases, they are not readily soluble, binding to fine grained soil particles, and particularly to organic matter (Pinoke, 1977). As a result, many pesticides are transported into water bodies primarily through soil erosion, rather than simply by runoff or drift. By the same token, these sorbed pesticides are released into the dissolved phase only slowly, but this does not mean that they pose no environmental threat. Bottom sediments are ingested by (and may accumulate in) benthic invertebrates, which in turn are consumed by fish. Incorporation into the vegetation of sloughs and wetlands may affect other forms of wildlife.

Hudson (1971, p. 297-99), in an early text on soil erosion, provided a short summary of the different kinds of pesticide in use at that time. Organo-chlorine insecticides (such as DDT) were regarded as the most serious because of their retention (and hence accumulation) in fatty tissue and the progressive buildup in the food chain. Since the early 1970's non-persistent insecticides have been used. Coote (1980) reported only occasional evidence of these newer forms in streams in southern Ontario.

Insecticides tend to fall into two main groups: organo-phosphorus compounds and organo-chlorines (or chlorinated hydro-carbons).

The former group, which includes parathion, malathion, and diazinon, break down relatively quickly, and for this reason tend not to be seen as a serious pollution problem, although acute toxicity is still a concern. In addition they have relatively high solubilities in water (Table 7.1) , though they still adsorb to soil particles. Gummer (1979) noted that, because of their rapid breakdown, they have gone largely undetected in the aquatic environment, and were not included in the WQB's routine monitoring program.

The organo-chlorine group (DDT, lindane (gamma-BHC), aldrin, dieldrin, heptachlor, endosulfan, methoxychlor etc.) are much more resistant to breakdown tending to accumulate up the food chain as already noted in the case of DDD. In general they have very low solubilities in water (except lindane) and sorb strongly to soil particles (Table 7.1). The use of organochlorines has gradually declined in the prairies, though toxaphene and chlordane were used in substantial amounts well into the 1980s, and, even today, lindane and methoxychlor remain in agricultural use (IEL, 1989).

In recent years, use of the carbamate group of insecticides (carbaryl, carbofuran etc.) has surpassed that of the organo-phosphorus compounds; the most extensive use has been that of carbofuran, especially for grasshopper control (Sheehan et al., 1987). The group degrades rapidly, however: monitoring of carbofuran in sediment and water after experimental spraying indicated no identifiable residue (Gummer, 1979), and it is not included in the regular WQB monitoring program.

In the case of herbicides, Hudson (1971) concluded that most of them did not appear to play a significant part in pollution resulting from erosion, because of their rapid breakdown in the soil, except for some of the copper and arsenic compounds used in the early days of herbicide application. Coote (1980) found evidence of herbicides in streams in agricultural parts of southern Ontario, but indicated no deleterious effects on fish.

In the context of the present study it is important to note that most acidic herbicide compounds (e.g. 2,4-D and MCPA which are widely used on the prairies for control of broadleaved weeds) have very high solubilities in water (Table 7.1) and therefore might not be expected to be significant in the particulate phase. As noted by Wauchope (1978), however, high solubility does not necessarily preclude strong adsorption by particulates, and the partitioning of these compounds between water and sediment does not seem to have been fully resolved: Gummer (1979) cites evidence for the sorption of both 2,4-D and MCPA by stream sediments. Development of accurate measurement of levels of acidic herbicides in sediments is an ongoing matter (Donald, 1990, pers. comm.), however, and currently WQB does not monitor this phase. Levels reported for unfiltered water samples are presumed to reflect the aqueous phase.

This high water solubility does not apply to some of the more recent non-polar herbicides, e.g. triallate and diclofop-methyl which are now used extensively for control of wild oats and form a major component of herbicide use in the prairies (Sheehan et al., 1987).

Fungicides were also regarded by Hudson (1971) as relatively minor hazards because of the limited quantities used and their application in small areas. A major exception is organo-mercury compounds applied as fungicide dressings to seeds.

Other organic compounds such as the polyaromatic hydrocarbons and PCBs, like pesticides, become tagged to fine-grained sedimentary particles and hence may be found in channel sediments. PCBs are common in farmland soils and are presumed to originate from atmospheric fallout (from industrial sources). Because they are bound up with soil particles, erosion of soils must be expected to be a major contributor of PCBs to channel sediments. Thus, while farming activity does not directly supply PCBs to streams, because of the extent of soil erosion on farmland slopes, PCBs need to be considered in any study of off-farm sediment impacts.

Though the deleterious effects of acute pesticide contamination are well known, there is not a great deal of information in many cases regarding the chronic toxic levels of the compounds. Saskmont Engineering Ltd (1989), for example, note that information regarding toxicity, and objectives or guidelines for the presence of compounds in water and, especially, sediment is lacking in most cases. Constable et al. (in press) provide a recent review on the use and application of pesticides in western Canada.

7.2 Ontario studies

Two studies were undertaken of levels of pesticides and PCBs in southern Ontario as part of the PLUARG study. Frank (1977) analyzed samples of suspended sediments from the mouths of the main rivers leading into the Great Lakes and found them all to contain DDT and/or its metabolites, averaging 138 ppb (ng/g) leading into Lake Ontario (1974) and ranging from 2 to 2380 ppb. Concentrations in streams draining to Lake Huron averaged 160 ppb in 1974 and 3 ppb in 1975. Levels averaged 34 ppb for streams draining into Lake Erie and Lake St. Clair (1974).

Dieldrin was found at most sites ranging in levels up to 14 ppb; endosulfan ranged up to 35 ppb; and PCBs ranged up to 1800 ppb, averaging 194 ppb for Lake Ontario, the most severely affected. No data were presented to show the relationship between levels of contaminants in the sediment and concentrations of sediment in the water.

Miles and Watson (1977) compared levels of contaminants in

bed material, suspended sediment and soil in the Big Creek watershed and Holland Marsh. In the former area levels of DDT averaged 4.0 ppm in the soil, but only 0.11 ppm in suspended sediment and 0.026 ppm in bed sediment. In the latter area, the equivalent values were 29, 8.0 and 0.3 ppm respectively. Although part of this decrease is related to the conversion of DDT into its less toxic metabolites over time, it would seem that levels in stream sediments are also appreciably diluted by mixing with non-contaminated sediments. Though the levels may seem small, the streamwater concentrations were still greater than the recommendations of the EPA in the USA.

Averaging a large number of samples from both areas, the sediment-associated contaminants accounted for 61% (ranging 30-93%) of the whole water (total) contaminant level in the case of the DDT group and 15% (ranging 3-42%) in the case of the more water-soluble dieldrin. Diazinon (which is more water-soluble than dieldrin) was not detected in the sediment phase at all, but found in water to levels higher than dieldrin. The preferential contamination of sediment, rather than water, is thus indicated to be largely controlled by the water-solubility of the compound, which varies from one group to another as already noted.

7.3 Organic compounds in the Saskatchewan R. basin

Several agencies have examined the problem in the Saskatchewan River basin: Environment Canada (Blachford and Ongley, 1984) and Alberta Environment (Hamilton and North, 1986) in the Bow and Oldman rivers; and Environment Canada (Ongley et al., 1985) and the Alberta Environmental Centre (AES, 1983a,b; 1984a,b) in the North Saskatchewan River; and Saskmont Engineering (1989) in the case of the South Saskatchewan River basin.

Gummer (1979) provided a review of pesticide monitoring in all three prairie provinces for the period 1971-1977. The residues monitored (determined in whole-water samples) are indicated in Table 7.2. Of these, the most commonly detected were the insecticides lindane and α-BHC and the herbicides 2,4-D (the most widely used by far), 2,4,5-T and 2,4-DP (dichlorprop). Figure 7.1 shows the pattern of detection of the first three, indicating peak detection of lindane and 2,4-D in the grain areas, while α-BHC is far more widespread.

The point was noted by Gummer (1979) that the lack of detection of most organochlorine residues in the surface waters of the prairies "can be explained for the most part by the reduction in use over the last few years .. and most importantly by the fact that these pesticides tend to accumulate in the sediment." The latter argument was also suggested for MCPA. This reference to sediment (usually viewed as bottom sediment) is an extremely important point in the context of the present review.

Gummer (1979) prefaced his report with the statement:

> 'Recognizing that sediments accumulate pesticides, the WQB chose however to monitor the water phase (including any suspended sediment) .... It was felt that any contamination of the aquatic environment would be at least detectable in the water phase, the phase more easily collected and analyzed."

This view was to some extent endorsed by Ongley (1982) who emphasized that in most Canadian rivers:

> "there is some doubt whether bottom sediments either reflect the geochemistry of suspended matter in transit above the bed, or characterize cumulative (time-integrated) stress over longer periods."

This is especially true in the Saskatchewan River basin where most of the bottom deposits are relatively coarse-grained, whereas pesticide residues preferentially attach to the silt and clay size fractions. Ongley (1982) went further, arguing that, notwithstanding the greater logistical convenience of "whole" water samples,

> "It is now beginning to be accepted that, for water quality purposes, suspended sediment must be explicitly examined rather than merely included within 'total' analyses of water samples."

Irrespective of whether contaminant studies use whole-water samples or suspended sediment samples, the recognition that contaminants are often concentrated in the suspended sediment phase has significant implications for the design and conduct of a water sampling program. WQB monitoring programs, for example, are generally based on sampling at fixed (usually monthly) intervals over time. In this situation there is a definite tendency to miss flood events, at which times sediment concentrations (and by implication contaminant concentrations in the whole water) are greatest. With this perspective, the temporal sampling representativeness of the data summarized by Gummer (1979) and the more recent data from IEL (1989), noted below, needs careful assessment.

An updating of Gummer's review was provided by Integrated Environments (IEL, 1989) based largely on the NAQUADAT database of Environment Canada and the ESQUADAT database of the Saskatchewan Department of Environment and Public Safety. The three most frequently detected residues in surface water were, again, α-BHC, lindane and 2,4-D. Others were 2,4,5-T, MCPA, atrazine, dichlorprop (2,4-DP) and picloram. The remaining 54 pesticides monitored were detected in less than 2% of the samples, 22 of them never having been recorded above their detection limits. Ignoring the acidic herbicide compounds which, based on their high solubilities in water, are not expected to be

significant in the sediment phase. the NAQUADAT and ESQUADAT databases indicate positive detection of the following pesticides in surface water:

> hexachlorobenzene, DDT and DDE, methoxychlor, heptachlor, endosulphans, chlordanes, alpha and gamma BHC, mirex, aldrin, endrin, dieldrin, dimethoate, triallate and atrazine;

the two BHC compounds were substantially higher in detection frequency than any of the others. Gummer (1979) noted that alpha-BHC is widespread in the Prairies, even in remote watersheds (including the Bow R. immediately below Banff National Park), from which he concludes that its presence must be caused by volatilization, atmospheric transport and redeposition. This may be true of many of the insecticides and should be borne in mind in interpretation of data describing geographic frequency of detection. High geographic detection, moreover, does not necessarily correspond to high concentration levels in specific areas.

The IEL report indicates elevated levels of pesticide residues in three areas: parts of the Qu'Appelle basin and southern Manitoba (both outside the present study) and the irrigation districts of southern Alberta. The results were depicted graphically in the form of maps of mean annual concentrations of the four main residues: lindane, alpha-BHC, 2,4-D and 2,4,5-T. One set is based on IWD watershed subbasins; a second set is based on ecodistricts of Environment Canada's Lands Directorate. The data from these maps are recompiled in Figure 7.2 which shows the locations of the areas with the highest levels of the four residues.

Figure 7.2 clearly indicates how little of the Saskatchewan River Basin aquatic ecosystem actually experiences high levels of pesticide residues. The exceptions (using the subbasin map) are:

lindane - at levels > 0.002 (to 0.008) ug/L in subbasins 5KC (mid-Carrot R.), 5KJ (Pasquia) and 5KK (part of the Saskatchewan River Delta lake area);

2,4,5-T - at levels > 0.02 (to 0.12) ug/L in subbasins 5AK (South Saskatchewan River area in Alberta), 5HA (Maple Creek), 5HB (around Leader Saskatchewan), 5GF (Sturgeon R.), 5GG (Prince Albert area), 5KE (Torch R. area) and 5KC (mid-Carrot R.);

2,4-D - at levels > 0.1 ug/L (to 1.0) in subbasins 5AG (around Taber, Alberta) and 5EC, 5EF, 5EG and 5GG (subbasins in the North Saskatchewan River system).

These values may seem relatively low: the lindane guideline given by CCREM (1987) for the protection of aquatic life, for example, is 0.01 ug/L. The level of 2,4-D thought to be detrimental to the inverterbrate community is about 1.0 ug/L

(Gummer, 1979). But these subbasin values are averages for relatively large areas and samples taken over several years.

Inspection of the ecodistrict map indicates the location of the high-level areas rather more precisely: in some cases only small parts of the subbasins exceed the stated concentration level. In addition there are some ecodistricts with high levels which are not indicated at the scale of the larger subbasins: an area around Sounding Creek in Alberta is shown with high levels of lindane.

The ecodistrict map is important in indicating that high levels of pesticide residue are restricted to small streams and rivers rather than being found in the mainstems as might at first be suggested by the watershed map. Neither map indicates how much of the residues were sediment-related, and it would be worthwhile reexamining the data in terms of concentrations per unit mass of sediment. The herbicide residues, based on the solubility data of Table 7.1, may have been largely in the dissolved phase. It is beyond the scope of the present study to investigate these high-level areas more closely.

In any case, it should not be assumed that all high-level sites reflect agricultural runoff. Gummer (1979) notes that lindane is registered in Canada not only for controlling ticks and flies on livestock, and seed treatment for wireworm control, but also for use in the logging industry. And though 2,4-D is primarily used for control of broadleaved weeds, the highest 2,4-D values on record in the prairies occurred at sites downstream of urban areas, again indicating that agricultural use is not the only source of pesticide residues in surface water.

The data sumarized by IEL (1989) were primarily from regular environmental monitoring programs. Special studies have been undertaken in selected parts of the basin, however, and some of these are indicated below.

7.3.1 The Bow and Oldman rivers

Though the Blachford-Ongley study covered organochlorine compounds, in most cases whole-water concentrations were below the level of detection or only slightly above. The percentage of the samples in the Bow-Oldman network that provided detectable levels of concentration are broken down by season and medium for different types of organochlorines in Fig. 7.3.

The preferential incorporation by sediment and plants, compared to water, is clearly indicated in most cases. The concentration of these substances in suspended sediment in streams is also shown to be much higher in summer and fall than in the spring. This was attributed to the higher percentage of organic material in stream sediment during the spring and fall (Fig. 7.4) and the preferential bonding of many of these substances to organic particulates.

No data are available for the loading of the organochlorines. It is possible that the same spring-time dominance in loading as found for phosphorus and metals also prevails for the organochlorines (even though most sediment at this time of the year appears to result from bank erosion), but this is conjecture.

The geographic distribution of the organochlorine data was not provided by Blachford and Ongley, and therefore no comment can be made regarding sources. The key question regarding the organochlorines, of course, is the effect of their presence on aquatic life, a question which remains largely unanswered in a quantitative way.

The Hamilton-North report reviews water quality data collected by Alberta Environment and Inland Waters Directorate during the 1970's on the Bow River. The study of trace organics included 23 compounds of which only five were detectable in water samples, and only two were observed regularly. These were the insecticide alpha-BHC (no longer used, but an impurity in lindane) and the herbicide 2,4-D. Levels of 2,4-D were low, the maximum being 0.6 ug/L which is less than the 4 ug/L CCREM (1987) guideline for aquatic life. It should be borne in mind that the review was restricted to water quality data (not sediment quality), and it is unclear whether samples were filtered or whole-water (and if the latter the extent to which the residues were bound up with the particulate phase).

7.3.2 North Saskatchewan River

Pesticide levels in the North Saskatchewan River have been examined by several agencies. Ongley et al. (1985) reported data for suspended sediments sampled during low-flow conditions in the summer of 1983. Though levels of DDT were not determined, aldrin ranged in concentration up to 16 ppb, and dieldrin up to 4.75 ppb (compared to 14 ppb in southern Ontario). Analysis of toxic response in bioassays provided inconclusive data: not all sites which showed toxic sediment contained detectable levels of EPA priority chemicals; and not all sites contaminated by priority chemicals showed bioassay toxicity in the sediment.

Data from the Alberta Environmental Centre (AEC, 1983a,b; 1984a,b) suggest that pesticide contamination of fish is not a serious issue in either the Red Deer River or the North Saskatchewan River, though levels were invariably larger in the latter than the former. The same study also showed that the concentration of PCBs in bottom sediments of the North Saskatchewan River was usually less than the detection limit of 0.01 ppb dry weight.

The WQB of Environment Canada has initiated a pilot program of suspended sediment sampling for pesticides on the North and South Saskatchewan Rivers using a passive sediment sampler

developed by B.C. Kenney at the National Hydrology Research Institute; no data are yet available.

7.3.3. South Saskatchewan River basin

In the South Saskatchewan River, toxicity of the sediments of Lake Diefenbaker has been assessed using algal and nematode (roundworm) bioassays (Gregor and Munawar, 1989). The tests indicated that the sediment chemical fractions frequently inhibit growth and maturation of the nematodes, with lethality being observed in 4 out of 12 sites. The cause of these physiological effects was not determined. They concluded that "additional controls need to be incorporated in the tests to determine the relative importance of the various contaminant groups (i.e. organic versus inorganic) as well as the role of nutrient availability in mitigating contaminant effects on the test organisms."

A detailed review of organic contaminants in the South Saskatchewan River basin was prepared by Saskmont Engineering Ltd (1989) for the South Saskatchewan River Basin Study. This review included documentation of different types of pesticides applied on farms, as well as data on contaminant levels in fish and sediment in the basin.

The review of permits issued by Saskatchewan Environment (Biocide Applications 1984-86) indicated pesticides for mosquito control to be a major source of organic compounds, especially in the Saskatoon area, as well as herbicides to control submergent and emergent aquatic vascular plants and algae. The 1986 Saskatchewan weed survey of cereal and oilseed crops provided data on insecticides and herbicides applied to fields in the area. In terms of insecticides, deltamethrin was the most widespread (on more than 40% of the area) followed by carbofuran (26%) and carbaryl (14%); all others were minor. In terms of herbicides, 2,4-D was applied on 50% of the area; bromoxynil, dicamba and MCPA on more than 20% of the area; with only 7% of the area untreated.

The Saskmont study also refers to the monitoring program by Environment Canada (Munro, 1987a) dealing with forage fish collected in the South Saskatchewan River at Highway 41 and at Lemsford and in Lake Diefenbaker. Alpha-BHC was found at levels of 9 ppb, DDT at levels of 1 ppb and PCBs at levels of 30 ppb. Another study by Environment Canada (Munro, 1987b) examined the presence and impact of pesticides in fluvial ecosystems in the basin as the result of chemical control of grasshopper infestations. Carbofuran was detected in the South Saskatchewan River at Highway 41 (0.0018 µg/L) and Swift Current Creek (0.0033 µg/L), but no mention has been found of the degree of contamination of sediments. Nicholaichuk and Grover (1983) and Nicholaichuk et al. (1987) monitored levels of herbicides in spring and irrigation runoff in the Swift Current Creek area.

Residues of the insecticide methoxychlor, which sorbs strongly to colloids (especially organic), were shown to exist in sediments collected from the South Saskatchewan River (Muir and Yarechewski, 1986), accounting for approximately half the total levels in whole water samples.

7.3.4 Tobin Lake

Sediment samples from Tobin Lake were also reported by Muir and Yarechewski (1986) as contaminated with methoxychlor (1.9-3.3 ppb), in contrast to the study by Sawchyn (1983) where the detection level was too high (50 ppb) to permit identification. The two researchers concluded that both sediments and sediment-dwelling organisms in the North and South Saskatchewan rivers were contaminated with low-levels of the insecticide throughout the year.

The toxicological significance of this contamination is a separate matter. Samoiloff et al. (1983) undertook biassay tests on chemically fractionated bottom sediments from two sites in Tobin Lake (along the mainstem axis) and found that, though strong toxicity existed in some fractions (one being semilethal to the organisms used), the most toxic fraction contained no detectable pesticides. And the fraction that would have been expected to contain pesticides showed no detectable toxicity to the organisms.

7.3.5 Battle River basin

Alberta Environment and Environment Canada are jointly involved in the monitoring of contaminants of various kinds in water, sediment and biota at more than six sites along the Battle River, Alberta (Crosley, 1990, pers. comm.). To date only one survey has been made of suspended sediment (using pump-centrifuge sampling system) and no data are yet available. About 40 samples of bottom sediment have been examined (though these tend to be sandy sediments in the downstream reaches). No organochlorine pesticide residues were found in these bottom samples (though concentrations were high in water samples). The only pesticide detected in the bottom sediments appears to have been the herbicide triallate.

7.3.6 Experimental plots

Nicholaichuk et al. (1989) monitored herbicide losses in drainage from a farm field irrigated with surface water using the corrugation method. The losses of herbicide in drainage water were in the range 0.1% to 0.7% of the amounts applied. Maximum concentrations were as follows: 2,4-D (3.5 µg/L), dicamba (4.7 µg/L), MCPA (12.9 µg/L), bromoxynil (20.4 µg/L) and 63.8 µg/L for diclofop. These peaks all occurred during the first flush of irrigation water. No data were provided for sediment concentrations in the runoff, and no breakdown was given for the particulate and dissolved components of the contaminants.

7.4 Conclusions

7.4.1 Sources and toxicity of contaminants

There are two distinct problems in the interpretation of most organic pollutant data in the aquatic environment of the Saskatchewan River basin. One is the determination of how much of the residue originates from local farmland (rather than industrial or municipal sources in the basin or atmospheric transport from outside). The second problem is the assessment of safe levels of contaminant in sediments and stream water, and the relationship between these two phases in the context of contamination of the food chain and water extracted for human consumption.

In terms of source, agriculture is certainly the single largest, but not the only, user of pesticides. For example, almost a quarter of the use of pesticides in Alberta in 1970-71 was apparently for urban home and garden use (AECA, 1973). Industry is also a significant user, especially the pulp and paper and seed dressing industries, as well as pesticide manufacturing and packing plants. Some of the highest pesticide residue levels have been detected downstream of urban areas and apparently associated with particular industrial plants (Gummer, 1979).

Regarding toxicity, samples to date seem to indicate only very low concentrations in bed sediments, while levels in fish, though higher, are considered safe. In terms of drinking water, it is the level of the dissolved phase rather than concentrations in sediment that matters. Analyses by the Water Quality Branch of IWD, Environment Canada show that most organics in the South Saskatchewan River are below detection limits except for alpha-BHC, lindane, atrazine and triallate. The role of sediment in sorbtion and subsequent release of these organic compounds is uncertain.

7.4.2 Role of sediment in transport of pesticides

Notwithstanding the recognition by WQB of the role of particulate matter in sorbing and transporting many pesticide residues, the reports and data currently available do not facilitate interpretation of the actual part played by fluvial sediment in pesticide contamination of watercourses in the prairies.

The IEL (1989) database, for example, differentiates between water and (bottom) sediments, but the water phase is apparently whole-water (including suspended sediment) in most cases. No analysis appears to have been undertaken of the concentration of sediment in the water (and its particle size), nor of the relative importance of the two phases in terms of the concentration of pesticide residues.

The reliability of the picture presented by these data is also unclear. This is especially true in terms of the representativeness of sampling times (and sampling sites) in the context of marked fluctuations in sediment concentration (and loading) during the open water season.

In short, before any accurate appraisal can be made of the role of sediment in pesticide levels and movement within watercourses, a detailed analysis of the sediment component of the existing pesticide water quality database is needed. This would require careful attention to such matters as the timing of sampling (in relation to the annual hydrograph), the location of sampling in a cross-section (given variability in levels, especially of the sand fraction, within the cross-section), the particle size of the sediment component, etc.

7.4.3 Emphasis on mainstem watercourses

Most of the data presented above pertain to the main stems in which pesticide contamination of water and sediments would be expected to be small because of the large quantities of "background" water (mostly from the headwaters) and sediment (assumed to be mostly from bank not surface soil erosion) carried in the mainstems. The matter may be quite different in smaller streams, draining only the prairies, such as the Battle River and Swift Current Creek. Though these water bodies have much less importance for fish and municipal supplies, it would seem important to undertake at least a reconnaissance examination of bottom and suspended sediments in some of these small streams that drain primarily cropland, as in fact has been started in the case of the Battle.

7.4.4 Isolated surface water bodies

The low levels of pesticide contamination of the mainstems may not be due entirely to the masking effect of uncontaminated sediment eroded from channel margins. It may also reflect the fact that a considerable amount of farmland soil simply does not reach the stream system (as indicated by the low sediment delivery ratios: Section 2), even in smaller basins.

The latter observation implies that the major environmental effects of sediment-associated contaminants from farmland are likely to be found either "on-farm" or in off-farm areas that are upstream of the drainage network (or simply not integrated with it at all). In the case of the hummocky terrain of the prairies, such an interpretation would seem to focus attention on internally-drained basins as the major recipients of farmland contaminants.

This point was emphasized by Sheehan et al. (1987) in their review of the impact of pesticides on the ecology of prairie nesting ducks. They note that the southern prairie belt of intensively cultivated agricultural land coincides with

productive waterfowl habitat for 16 million ducks which breed annually in the sloughs (or "potholes") of the hummocky landscape: and that more than half of these ducks nest in close association with current agriculture. With a slough density averaging about twelve per square kilometre (most being smaller than 0.2 ha), this is not a minor issue.

To the extent that these sloughs are not part of the integrated drainage network, the problem is beyond the terms of reference of the present study. In addition, it is unclear how much of any pesticide contamination results from drifting during spraying rather than from transportation into sloughs by surface water flow and sediment movement. The issue is not pursued here except to emphasize (following Sheehan et al., 1987) that, while legitimate concerns exist regarding the impact of pesticides on waterfowl in the prairies, the paucity of information currently available impedes an accurate evaluation of the problem.

One of the recommendations of the Sheehan report was that "water monitoring agencies such as the Inland Waters Directorate should be encouraged to monitor pesticide residues in slough waters". In fact, WQB has already initiated a multi-phase (spring runoff, sediment, aquatic invertebrates) investigation of pesticide residues in terminal prairie lakes. The program involves comparison of fifteen "dry lakes" (resulting from the long drought of the last decade and the extreme drought of 1988) and ten permanent lakes. Complete results are not yet available (Donald, 1990, pers. comm.).

8. CONCLUSIONS

Two separate questions need to be posed in this conclusion:

(8.1) What are the impacts of stream sediments and sediment-bound contaminants in the Saskatchewan River basin?

(8.2) How much of the stream sediments originates from farmland erosion and, therefore, how much of the sediment impact can be attributed to farming?

8.1 Sediment impacts in the Saskatchewan River basin

Generally speaking, sediment impacts in the river basin appear to be small.

Suspended sediment concentrations are generally low. Turbidity appears to have had little impact on fish; in-stream use of water for generation of electric power (or for cooling of thermal plants) has not been noticeably affected, though there have been sediment problems with the cooling water in the Souris River basin; high turbidity itself appears to pose little threat to recreational activity (but see comments regarding eutrophication); and no problems have been identified with off-stream use of water for municipal and industrial needs (though no data have been found for the costs of filtration).

Sediment accumulation (sedimentation) does not appear to have been reported as a problem on watercourses of the river network. There are, however, very few data at the scale of roadside ditches, irrigation canals and small reservoirs, and the conclusion may not be applicable at this scale.

Eutrophication is rarely a problem on the larger watercourses (except downstream of sewage plants), but is often regarded as the main water quality issue in the region in the context of small water bodies. In some cases this is the result of industrial activity or cottage development, but there is definite evidence to implicate sediment as well. High total phosphorus levels in spring and early summer are the result of high concentrations of suspended sediment (to which P is bound) and high late summer levels have been documented to result from P release from bottom sediments.

Metal concentrations in fluvial sediments do not appear to pose a problem in most cases, though most water quality data are for in-water concentrations rather than in-sediment levels. Aluminum, lead and zinc occur (primarily associated with fine suspended sediment) in the South Saskatchewan River basin at levels that are possibly marginally toxic to aquatic life. Iron and manganese were similarly noted to occur at high levels and in association with clay size sediment, though the impact here was essentially aesthetic. Mercury is found at relatively high

levels in bottom sediments and also in fish, though the levels vary from year to year.

Organic contaminants have been detected in the surface waters of the basin and are often sorbed to sediment depending on the in-water solubility of the compound. These include HCH, DDT and its metabolites, the herbicide 2,4-D and methoxychlor residues, as well as PCBs. The levels are generally not regarded as serious, but there is some debate as to exactly what levels can be regarded as acceptable. Bioassay tests using sediments from the North Saskatchewan River and from Lake Diefenbaker certainly show varying degrees of toxicity, but the cause of this toxicity has not yet been determined. Most of the data so far available are for large rivers.

In summary, direct sediment pollution (arising from high concentrations and loads of suspended sediment) does not appear to be a problem in the Saskatchewan River basin. Impacts resulting from contaminants associated with sediment also do not appear to be a problem on the larger watercourses, except mercury in some cases. Problems do exist in smaller water bodies, especially eutrophication resulting from sediment-associated phosphorus, and may also occur with organic contaminants, but few data are available in the case of the latter.

8.2 Off-farm impacts in the Saskatchewan River basin

Though sediment concentrations and loads are generally low throughout the basin, what is perhaps more significant is that the proportion of sediment from farmland erosion seems to be relatively insignificant. This certainly appears to be true for the mainstem rivers, and, for this reason, whatever the sediment impacts on the larger rivers (which have just been noted to be small), the contribution of farmland erosion to these impacts is likely to be minimal.

The impact of farmland erosion is much greater on smaller watercourses, not because sediment yields from farmland are higher in small basins, but because there is less sediment from other sources to "dilute" the farmland sediment to background levels. The sediment-associated eutrophication in small waterbodies appears to be directly related to farmland erosion. This is not because farmland soils are particularly enriched in phosphorus relative to non-farmed soils and sediments (they are not), but rather because surface soil erosion (in contrast to bank erosion and gullying) selectively removes the fine grained sediments and organic complexes which incorporate most of the phosphorus.

It would be expected that organic contaminants, such as pesticide residues and PCBs, would occur at higher levels in stream sediment from small farmed catchments, but few data on organic compounds have been found in small basins. Some care is also required in interpreting such data. PCBs are not

intentionally applied to farmland soils, but occur in surface soil from atmospheric fallout. Some pesticides may similarly be dispersed by atmospheric processes so that the residues found in small streams may be unrelated to land use in their watersheds.

This point should be emphasized. In terms of sediment-related phosphorus (but not dissolved), almost all the P lost from farmland fields to streams is phosphorus that is naturally found in the soil (both inorganic and organic) and not the result of fertiliser applications. Streams become enriched with nutrients in farmed watersheds simply because of the selective removal of phosphorus-containing sediment. In the case of PCBs (just like Cs-137), stream sediments are likely to be enriched in farmland areas, not because farmland soils themselves are enriched, but because surface soils are enriched, and farmland erosion (unlike bank erosion and gullying) affects surface soils. The same may be true of organic compounds from pesticide residues and other contaminants: soil erosion may be a problem not so much because of high levels of residues in farmland soils compared to native soils, but because of contamination of surface soils and the preferential erosion of surface soil on farms.

Thus it is not so much that farming itself produces sediment-bound contaminants as that farmland erosion, being a surface soil erosion process, produces higher levels of these pollutants than other forms of erosion which act primarily on subsurface sediments. The information documented in this report certainly implicates farmland erosion in nutrient-contamination of small waterbodies. Whether farmland erosion also produces organic contamination (PCBs, pesticides) cannot be answered at the present time, partly because of lack of data in small basins, and partly because of uncertainty as to the level at which these compounds become pollutants.

8.3 Comparison with other parts of Canada

In comparing the information obtained for the Saskatchewan River network with other parts of Canada, two questions require answers.

i. Why is it that the impact of agriculture on fluvial sediments in the Saskatchewan River basin is so much less than reported for southern Ontario?

ii. How can the contrasts between southern Ontario and the Saskatchewan River basin be used to extrapolate these findings to other parts of Canada such as the Maritimes?

No definitive answer can be provided to the first question, but clearly it is to be found in one or both of two points: either sediment-associated contaminants are produced from farmland at higher levels of yield in southern Ontario basins and/or the farmland sediment found in streams of the Saskatchewan

River basin is simply diluted more with uncontaminated background sediment.

No attempt has been made in this study to address the first possibility, but provincial data for on-farm soil erosion costs indicate that, on a per hectare basis, Alberta and Saskatchewan are only slightly behind Ontario (Table 8.1). In terms of the second possibility, several aspects of the fluvial sediments in the two regions warrant comment.

In the first place there is a major contrast in size between the PLUARG basins in Ontario and those for which sediment data are available in the Saskatchewan River basin: the PLUARG basins ranged from 18 to 79 km^2; most of the Saskatchewan River basin data are for the main stems with much larger basins. There is a general tendency in much of Canada for sediment yields and concentrations to increase downstream as more and more riparian sediment is incorporated into the suspended load. Thus bigger basins will contain a much greater percentage of "background" riparian sediment capable of diluting (masking) contaminants from agriculture. In the PLUARG study, cropland erosion accounted for generally close to 90% of suspended sediment in streams at the basin mouths. The figure for sediment stations in the Saskatchewan River basin is probably an order of magnitude smaller.

To some extent, the contrast between the two data sets may be an artifact of sampling: the PLUARG data set was biased to the smaller basins in southern Ontario, while little information is available for small tributary streams in the Saskatchewan River basin data set. It is true that, in general, basins are much smaller in southern Ontario than in the Prairies, but the data currently available simply do not allow a valid comparison between basins of the same size in the two areas.

In any case, there is a second point which differentiates the two regions. There is much greater connectivity between farmland slopes and the stream network in southern Ontario than in the Prairies. Much more of the sediment that is removed from fields would be expected to reach the stream system in southern Ontario than would be the case in the Saskatchewan River basin. The latter is made up of rolling terrain in which much of the land has no surface drainage connection with the stream system. This is a second reason, therefore, why a much larger percentage of stream sediments in Ontario would be expected to originate from farmland than in the Prairies.

There may be a third consideration too. Fluvial sediments appear to be much coarser in the Saskatchewan River basin than in most of the basins of southern Ontario. This is certainly true of the bottom sediments which are mostly gravel in the Alberta parts of the basin and sand through Saskatchewan. In contrast, many of the streams of southern Ontario are sand-bedded or even muddy. It is more difficult to compare suspended sediments

without the processing of large quantities of data, but preliminary inspection of WRB data suggests the same comment holds in this case too. Suspended sediment along the main channels of the Saskatchewan River basin (Oldman, Red Deer, South and North Saskatchewan rivers) typically contains 20 to 40% sand. This seems to be higher than southern Ontario, though data for only a few of the PLUARG basins have been found; these indicate very low sand-content levels, generally less than 10 percent (Smith, 1990, pers comm.). Irrespective of source, therefore, Prairie stream sediments may be "cleaner" because of a larger percentage of coarser material deficient in adsorbed contaminants.

All these points should be borne in mind in attempting to extrapolate these data to other parts of the country. In the Maritimes, for example, basins are generally much smaller than in the Saskatchewan River system, and more comparable with southern Ontario. Sediment delivery ratios may be higher in the Maritimes because of better integration of stream networks. Sediment concentrations are likely to vary considerably according to soil type, terrain conditions, and bedrock geology, but sediment-associated contaminants from farmland probably occur at much higher levels there than in the Prairies. The impacts of this sediment may also be much greater in the Maritimes with the accumulation of sediment in the estuaries of the region, the setting for much of the recreational and economic activity.

Finally, it would be unwise to extrapolate these conclusions to the rest of the Prairies. As shown in Fig. 2.1, farmland erosion appears to be much more severe in <u>other</u> parts of the Prairies, especially in the Peace River district and tributaries of the Milk River system. Off-farm impacts are probably much more evident in these areas also. It is appropriate to end with the comments of Anderson and Knapik (1984, p. 43):

"Water erosion on glacial landscapes with poorly integrated drainage systems, which characterize much of the Prairies, results in redistribution of top soil in a small area, often within the same field. This results in less damage to neighbours fields and public structures, and less sedimentation of river systems than in landscapes with long planar slopes and integrated drainage systems. The Peace River region is somewhat of an exception to this pattern, with many long planar slopes and examples of off-farm damages by water erosion which are quite visible."

9. RECOMMENDATIONS

The major deficiency in the database relating to off-farm sediment impacts in this region is the scarcity of data from small drainage basins. These are the watersheds in which the proportion of sediment from farmland sources is likely to be highest, and where the impacts on sediment and water quality might be expected to be more apparent.

There are two avenues of investigation that might be followed in order to address this deficiency. One approach is through an inspection, analysis and evaluation of the existing WRB sediment data for small-stream miscellaneous sediment stations (Section 2.6, p.15, and App. 2) and WQB whole-water data from small basins, as discussed in Section 7.4.2 in connection with pesticides. The other is through a new phase of field monitoring in which greater emphasis is placed on small basins.

The first two recommendations deal with further analysis of the existing database; the second two relate to any new fieldwork phase dealing with small basins.

Recommendation 9.1

Sediment data collected by WRB at miscellaneous stations for small basins in the prairies, and not previously analyzed by Ashmore (1986) (whose review focussed primarily on full-program stations and large basins) should be examined further to assess the causes in inter-basin variability in sediment yield. The work should attempt to quantify the proportion of sediment originating from farmland sources in high-yielding basins.

Recommendation 9.2

Pesticide data for whole-water samples (and which include suspended solids concentration or non-filterable residue) taken at the mouths of small prairie basins by WQB should be investigated with a view to determining the relationship (if any) between high contaminant levels and high concentrations of suspended sediment. Attention will need to be directed to the sampling representativeness of the sediment data as indicated in Conclusion 7.4.2.

These data should then be examined in an attempt to determine the factors causing inter-basin variance in sediment-associated pesticide levels, with special reference to sources of pesticide (e.g., agricultural, municipal, industrial) and sediment (bank erosion, farmland erosion etc.).

Ideally, these two studies should be initiated soon so that the results of the studies could be used by WRB and WQB in setting up any new small-basin field program dealing with sediment quantity and quality. The area of investigation of the two studies should be expanded beyond the boundaries of the

Saskatchewan River basin to include the full area of the prairies; this would seem to be appropriate given the higher contaminant levels found in many small basins outside the Saskatchewan River basin.

There may, however, be some incentive for WRB and WQB to become involved in small-basin monitoring of sediments before these data-review studies are complete. The National Soil Conservation Program of Agriculture Canada, for example, is in the process of initiating research into sediment movement off farmland in different areas across the country (Coote, 1990, pers. comm.). The work will be a subcomponent of the Soil Quality Evaluation Program and will be an integral part of both the Water and the Wind Erosion Monitoring Prediction Studies (WAMPS and WIMPS). Attempts will be made to measure off-farm transport of soil and associate chemicals at benchmark sites across the country and to extrapolate the results to larger areas of farmland. These studies are supposed to start this summer, and will run for 3 years under NSCP funding, with critical activities being continued for approximately another 5 years as part of the Land Resource Research Centre's regular program (Coote, 1990, pers. comm.).

The work described above concerns monitoring of sediment movement at the field scale. There would seem to be excellent opportunity for Inland Waters to collaborate with Agriculture Canada in this endeavour. IWD hydrometric monitoring and water sampling in the same areas, but at the watershed scale, would allow the two sets of data to address much more carefully questions relating to the effects of farmland erosion at sites further downstream away from fields.

A major difficulty with present IWD data lies in quantification of sources of supply of sediment and associated contaminants at IWD sites. The collaboration just described would go a long way to reducing this uncertainty.

Recommendation 9.3

Appropriate personnel of Inland Waters (WRB and WQB) should contact personnel of Agriculture Canada (NSCP) in order to ascertain the current status of the Soil Quality Evaluation Program and to explore the merits and costs of setting up stream sampling and hydrometric monitoring in some of these targetted areas.

Recommendation 9.4

In the event that IWD involvement in WAMPS and WIMPS is not possible, small-basin field studies might be implemented by IWD, guided by the results of 9.1 and 9.2. It would be sensible to seek the active participation of other agencies in these studies so that in-basin (sediment source) research (e.g. Cs-137 determinations) would supplement IWD monitoring.

All four recommendations above deal primarily with suspended sediment. Very few data appear to be available with regard to contaminant levels in the bottom sediments of small prairie streams. Integrated Environments Ltd (1989, p. 40) reviews some of these data. It concludes: "Records of pesticide sampling in sediments and biota are extremely sparse both inside and outside the NAQUADAT data base. IWD should initiate at least a limited sampling program for selected pesticides in these two components." The same recommendation, subject to certain caveats, is made here.

Recommendation 9.5

Sampling of bottom sediments and analysis for agriculture-related contaminants should assume an increasing proportion of WRB-WQB fieldwork in the future.

Data from such sediments could be used for three purposes:

1. as an alternative approach to the detection of contaminants in the environment, supplementing the regular water quality monitoring program;

2. as part of the overall diagnostic investigation in case studies where biota are known to be contaminated;

3. as a supplement to small-basin suspended sediment sampling programs (9.4) to assist in the interpretation of sources and fate of contaminants in suspended sediment.

Nonetheless it should be recognized that many of the bottom sediments of Prairie rivers are relatively coarse grained (especially along the main stems) and unlikely to show significant contamination, given the propensity for contaminants to sorb to fine-grained sediments. Moreover even fine-grained beds may be inappropriate where the bed material is of geologic age (e.g. pro-glacial lacustrine) rather than modern alluvium. Prior determination of the physical character of bottom sediments is essential to assist in a cost-effective analysis of contaminants in these sediments.

The hydrological setting must also be borne in mind in such a sampling program, just as in a water quality monitoring program. It has been suggested throughout this report that suspended sediments in small watercourses are likely to be much more contaminated than those of larger rivers because of the "dilution" effect of contaminant-free sediment scoured from channel margins by the mainstem streams.

A similar argument would prevail in terms of bottom sediments. Large reservoirs such as Lake Diefenbaker and Tobin Lake are logical places for study, but, while effectively integrating the suspended load from upstream, are likely to have lower diffuse-source contaminant levels in bottom sediments than

lakes and reservoirs on small streams. As an example, it has already been noted that sediments in the "polluted" part of Tobin Lake have lower metal concentrations than in the side bay receiving flows from the small Petaigan river.

The attention to small prairie watersheds recommended above is not meant as a criticism of past sampling strategies. These were dictated by the mandates of the various branches, and by the natural concerns regarding the water quality of the main stems. The existing sampling program does not, however, adequately cater to the specific issues addressed in this report.

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APPENDIX 1

METHODS USED IN THE CONSTRUCTION OF THE 1:1 MILLION SOIL EROSION RISK MAPS

(Sections 2.3 and 2.4)

Wind erosion

$$E = K\ C\ (V^2 - mW^2)^{1.5}$$

where
- E = maximum instantaneous soil movement;
- K = surface roughness and aggregation factor;
- C = factor representing soil resistance to movement by wind;
- V = drag velocity of wind at soil surface;
- m = moisture effect on shear resistance;
- W = wetness or moisture content of the surface soil

Values of K, C and m were initially selected for a small number of soil types from published research results. The values for other soils were then estimated from their clay content and from the expected behaviour of each surface texture class when compared with the soils used in the reported research. The m term is a constant at all sites; the K and C terms vary according to the soil texture group.

The wetness of the surface soil (W) was based on the assuption that the wheat-summer fallow rotation is used, one in which the spring period (April-May) preceding planting is taken as the critical period for soil erosion. Long-term data for temperature, precipitation, potential evapotranspiration and crop growth were then used for purposes of predicting the available moisture in the surface soil (the top 5 cm) for each of four soil texture classes for the first day of the month in April, May and June. An average value of these months was then computed for each of the four texture groups. These were plotted for all weather stations, isolines drawn on separate maps for each texture group, and the appropriate group used for each polygon of the soil landscape map.

The wind speed index (V) was based on values of the mean maximum 1-hr windspeeds during April and May for all the climate stations with these data, and the average value taken for those two months. In southern Alberta, however, a zone was identified as being particularly susceptible to winter chinook winds: within this zone, mean maximum 1-hr windspeeds were calculated for January through May rather than only for April-May. All speeds were adjusted to a common height of 2 m above ground level. Isolines of this windspeed were then drawn and a single representative value estimated for each polygon of the soil

landscape map. The drag velocity was then obtained from the equation employed by Chepil and Woodruff (1963).

The values of the dimensionless index E were classified as negligible (<100), low (100-250), moderate (250-400), high (400-700) and severe (>700).

The proportion of each polygon area under cultivation was determined from Landsat satellite imagery during the growing seasons of 1980 to 1983.

The degree of protection provided by the usual crop management practice within the cultivated area was determined on the basis of the quantity of crop cover and residues (kg/ha) expected in April and May, together with regression constants and coefficients that varied according to the crop group.

Water erosion

$$A = R \times K \times LS \times C \times P$$

where A is the predicted soil loss from water erosion (t/ha/yr), R is the erosivity of rainfall and snowmelt; K is a soil erodibility factor, and LS is a dimensionless slope length and steepness factor.

The terms C and P denote dimensionless measures of crop management (a ratio which compares the soil loss with that from a cultivated bare fallow field) and conservation practices (a ratio which compares soil loss with that from a field with no conservation practice i.e. ploughing up and down slopes) respectively. In producing these soil risk maps, fields were assumed to be bare (C=1.0) and to have no conservation practices such as terraces or contour strip cropping (P=1.0).

Erosivity of the rainfall was estimated using the maximum 6-hr rainfall expected once every two years; and erosivity of snowmelt was estimated using the procedure of McCool et al. (1982) as modified for conditions in Alberta by Tajek et al. (1985) and based on the water equivalent of the average amount of snow on the ground at the end of March.

Soil erodibility was estimated for each polygon using the equation of Wischmeier and Smith (1978), based on percentage of clay and organic matter in the soil, soil structure and soil permeability. It is worth emphasizing that whereas loams (especially clay loams) are rated least erodible in terms of wind erosion (the combined KC factor in the WEE), loams are rated most erodible in terms of water erosion.

APPENDIX TWO

MISCELLANEOUS SEDIMENT STATIONS IN SASKATCHEWAN RIVER BASIN FOR WHICH SEDIMENT LOADS COMPUTED

(Four pages)

Identification of sediment stations for which suspended load computations were undertaken using sediment rating curve technique (two pages);

and estimates of mean seasonal (April-October) suspended sediment yield based on sediment rating computations (two pages).

Data from Ashmore (1986) Tables 5.16 and 5.19.

TABLE 5.16
Sediment Stations for Which Rating Curve Load Computation was Carried Out

1. Miscellaneous Stations

Station Number	Station Name
05AA004	Pincher Creek at Pincher Creek
05AA011	Mill Creek near the mouth
05AA022	Castle River near Beaver Mines
05AA026	Dutch Creek near the mouth
05AA027	Racehorse Creek near the mouth
05AA028	Castle River at Ranger Station
05AB013	Beaver Creek near Brocket
05AB029	Meadow Creek near the mouth
05AD002	Belly River near Stand Off
05AD028	Waterton River near Glenwood
05AE002	Lee Creek at Cardston
05AE027	St. Mary River at International Boundary
05AE037	Lee Creek at Beazer
05AE039	Tough Creek near Beazer
05AJ001	South Saskatchewan River at Medicine Hat
05BH009	Jumpingpound Creek near the mouth
05BK001	Fish Creek near Priddis
05BL007	Stimson Creek near Pekisko
05BL009	Highwood River near Aldersyde
05BL013	Threepoint Creek near Millarville
05BL014	Sheep River at Black Diamond
05BL019	Highwood River at Diebel's Ranch
05BL021	Highwood River below Picklejar Creek
05BL022	Cataract Creek near Forestry Road
05BL023	Pekisko Creek near Longview
05CA002	James River near Sundre
05CA009	Red Deer River below Burnt Timber Creek
05CB002	Little Red Deer River near Water Valley
05DA002	Siffleur River near the mouth
05DB001	Clearwater River near Rocky Mountain House
05DC002	North Saskatchewan River at Saunders
05DC006	Ram River near the mouth
05DF001	North Saskatchewan River at Edmonton
05EC002	Waskatenau Creek near Waskatenau
05EC003	Redwater River near Redwater
05EE003	Vermilion River near Vegreville
05EE004	Vermilion River near Hazeldine
05EE007	Vermilion River near Marwayne
05HD036	Swift Current Creek below Rock Creek
05KC002	Connell Creek near Connell Creek
05KH009	Saskatchewan River Old Channel

TABLE 5.16 (cont'd)
Sediment Stations for Which Rating Curve Load
Computation was Carried Out

1. Miscellaneous/Seasonal Stations

Station Number	Station Name
05AA008	Crowsnest River at Frank
05AA023	Oldman River near Waldron's Corner
05AB021	Willow Creek near Claresholm
05AB028	Willow Creek above Chain Lakes
05BA002	Pipestone River near Lake Louise
05BC001	Spray River at Banff
05BH004	Bow River at Calgary
05BJ004	Elbow River at Bragg Creek
05BL024	Highwood River near the mouth
05BM004	Bow River below Bassano Dam
05DC001	North Saskatchewan River near Rocky Mountain House
05FE004	Battle River near the Saskatchewan Boundary
05FF001	Battle River at Battleford
05GC006	Eagle Creek near Environ
05HD039	Swift Current Creek near Leinan
05KB003	Carrot River near Armley
05KB005	Burntout Brook near Arborfield
05KB006	Leather River near Star City
05KB011	Doghide River near Runciman
05KC001	Carrot River near Smoky Burn
05KD003	Saskatchewan River below Tobin Lake

3. Seasonal Stations

05AB030	Streeter Creek (main stem) near Nanton
05BB003	Forty Mile Creek near Banff
05CA003	Deer Creek (main stem) near Sundre
05KD001	Saskatchewan River at Nipawin

TABLE 5.19
Mean Seasonal Sediment Yield at Miscellaneous Stations

Station Number	"Effective" drainage area (km^2)	Sediment Contributing area (km^2)	Estimated mean seasonal load (tonnes)	Estimated mean seasonal yield ($t\ km^{-2}a^{-1}$)
05AA004	155	155	5 796	37.4
05AA011	179	179	8 457	47.2
05AA022	826	826	31 761	38.4
05AA026	143	143	2 151	15.0
05AA027	217	217	6 550	30.2
05AA028	376	376	18 662	49.6
05AB013	256	256	7 912	30.9
05AB029	130	130	8 936	68.7
05AD002	1 135	1 135	211 470	186.3
05AD028	1 600	344	31 800	92.4
05AE002	307	307	68 105	221.8
05AE027	1 165	1 165	96 308	82.7
05AE037	175	175	13 168	72.2
05AE039	39	39	246	6.4
05AJ001	42 112	26 384	2 887 642	109.4
05BH009	571	571	2 487	4.4
05BK001	259	259	1 550	6.0
05BL007	263	263	4 967	18.9
05BL009	2 337	2 337	97 786	41.8
05BL013	507	507	21 075	41.6
05BL014	595	595	40 742	68.1
05BL019	774	774	14 147	18.3
05BL021	132	132	1 560	11.8
05BL022	166	166	3 813	23.0
05BL023	231	231	7 668	33.2
05CA002	821	821	174 012	211.9
05CA009	2 247	2 247	36 356	16.2
05CB002	449	449	787	1.75
05DA002	510	510	3 219	6.31
05DB001	3 208	3 208	180 247	56.2
05DC002	5 158	3 894	125 455	24.3
05DC006	1 860	1 860	245 533	132.0
05DF001	27 336	17 322	4 401 422	254.0
05EC002	205	205	336	1.64
05EC003	1 060	1 060	4 197	4.0
05EE003	613	613	2 182	3.6
05EE004	3 573	1 573	30 752	8.6
05EE007	3 106	1 106	3 715	1.2
05HD036	1 086	1 086	2 542	2.3

TABLE 5.19 (cont'd)
Mean Seasonal Sediment Yield at Miscellaneous Stations

Station Number	"Effective" drainage area (km^2)	Sediment Contributing area (km^2)	Estimated mean seasonal load (tonnes)	Estimated mean seasonal yield ($t\ km^{-2}a^{-1}$)
Miscellaneous/Seasonal Stations				
05KC002	353	353	4 607	13.0
05KH009	-	-	60 610	-
05AA008	402	402	3 990	9.9
05AA023	1 444	1 444	16 699	11.6
05AB021	1 102	1 102	28 974	26.3
05AB028	162	162	2 620	16.2
05BA002	306	306	1 535	5.0
05BC001	749	632	3 939	6.2
05BH004	7 816	1 264	29 073	23.0
05BJ004	792	792	12 810	16.2
05BL024	3 982	3 982	56 534	14.2
05BM004	18 120	10 340	154 190	-
05DC001	10 983	3 894	1 358 678	123.7
05FE004	10 841	6 620	95 408	14.4
05FF001	13 616	9 394	1 245 546	132.6
05GC006	2 094	2 094	8 204	3.9
05HD039	2 596	858	5 033	5.9
05KB003	2 443	2 443	66 499	27.2
05KB005	285	285	11 079	38.9
05KB006	122	122	711	5.83
05KB011	434	434	16 936	39.02
05KC001	7 121	7 121	178 443	25.06
05KD003	167 612	-	98 073	-
Seasonal Stations				
05AB030	6.0	6.0	32	5.30
05BB003	133	133	439	3.30
05CA003	5.4	5.4	16	2.96
05KD001	166 179	166 179	7 698 681	46.33

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Summary of Damage Costs
(million 1980 dollars)

Type of impact	Range of estimates	Single-value estimate	Cropland's share
In-stream effects			
Biological impacts		no estimate	
Recreational	950-5,600	2,000	830
Water-storage facilities	310-1,600	690	220
Navigation	420- 800	560	180
Other in-stream uses	460-2,500	900	320
Subtotal—In-stream (rounded)	2,100-10,000	4,200	1,600
Off-stream effects			
Flood damages	440-1,300	770	250
Water-conveyance facilities	140- 300	200	100
Water-treatment facilities	50- 500	100	30
Other off-stream uses	400- 920	800	280
Subtotal—Off-stream (rounded)	1,100-3,100	1,900	660
Total—all effects (rounded)	**3,200-13,000**	**6,100**	**2,200**

Source: Original Conservation Foundation research.

TABLE 1.1 OFF-FARM IMPACTS OF ERODING SOILS IN THE USA
(from Clark et al., 1985)

TABLE 1.2 (from ESCC. 1983)

ESTIMATED ANNUAL OFF-FARM COST
OF AGRICULTURAL LAND DEGRADATION IN ONTARIO[1]

	Cost (Millions of $) ESCC (1983)	1984$[2]	% Attributed To Agriculture	Agriculture's Share (1984$)
Dredging Sediment from Harbours (Federal Program)	8.9	9.3	85[3]	7.9
Sediment Damage to Inland Lakes, Reservoirs, Channels	3.2	3.3	85[3]	2.8
Water Treatment Costs	0.5	0.5	85[3]	0.4
Sediment Removal from Road Ditches	3.6	3.8	50[4]	2.0
Sediment Removal from Municipal Drains	7.2	7.5	50[4]	3.8
Recreational Fisheries Losses	83.7	87.4	85[3]	74.3
TOTAL	$107.1	$111.8	-	$ 91.2

1 Covers only those off-farm costs estimated by the Provincial Erosion and Sedimentation Control Committee in 1983

2 1983 cost multiplied by the Consumer Price Index for 1983-84 (1.044)

3 Basis: Wall et al. (1982) finding that 70 - 100% of suspended sediment in agricultural watersheds was contributed by cropland erosion. The majority of erosion losses in Ontario are expected to occur on agricultural watersheds.

4 A large portion of sediment found in municipal drains and road ditches comes from bank erosion. A study conducted in the South Nation River Basin (Water & Earth Science Associates, 1981) showed that agricultural land erosion accounted for approximately 2% of municipal drain sediment; however, ditch banks are very unstable in this area. Another study in Southwestern Ontario found that 50% of sediments in municipal drains came from field erosion (Culley and Bolton, 1983). Other studies assume 100% of sediment in ditches and drains comes from field erosion (e.g. U.S. study reported by Wall and Dickinson, 1978). A value of 50% was chosen for this study.

Watershed	Mean stream sediment yields† ($kg \cdot ha^{-1} \cdot yr^{-1}$) 1975–1977	1976	1976 streambank erosion estimates ($kg \cdot ha^{-1} \cdot yr^{-1}$)	1976 cropland sediment yield estimates§ (%)	Streambank yield as proportion of total sediment yield (%)	Cropland yield as proportion of total sediment yield (%)
AG-1	961	998	223	775	22	78
AG-2	153	140	10	130	7	93
AG-3	197	258	24	234	9	91
AG-4	464	419	137	282	33	67
AG-5	274	351	5	346	2	98
AG-6	60	64‖	10	54	16	84
AG-7	98	43	7	36	16	84
AG-10	300	375	17	358	5	95
AG-11	—	19‖	65	—	—	—
AG-13	499	310	41	269	13	87
AG-14	139	135	75	60	—	—

*Data source: IJC–PLUARG Canadian agricultural watershed studies.
†Using NAQUADAT method of sediment yield computation (Demayo and Hunt 1975).
‡Knap *et al.* (1979).
§1976 stream sediment yields minus 1976 streambank erosion estimates.
‖Problems with streamflow measurement account for the very low sediment yield.

TABLE 1.3 SEDIMENT SOURCES IN THE PLUARG WATERSHEDS (from Wall et al., 1982)

TABLE 1.4 (from Ashmore, 1986)

MAJOR ONSTREAM STORAGE RESERVOIRS IN THE SASKATCHEWAN RIVER BASIN

Dam	Reservoir	River	Year	Capacity ($dam^3 \times 10^{-3}$)	Purpose
NORTH SASKATCHEWAN RIVER SUB-BASIN					
Bighorn	Abraham Lake	N. Saskatchewan	1972	1 768[1]	Hydro
Brazeau	Brazeau	Brazeau	1961	712[1]	Hydro
Coal L.	Coal L.	Battle	1973	25[3]	Flow augmentation
Driedmeat L.	Driedmeat L.	Battle	1974	15[3]	Recreation
Forestburg	Forestburg	Battle	1956	28[2] 15[1]	Cooling water
RED DEER RIVER SUB-BASIN					
Dickson	Glenifer L.	Red Deer	1983	203	Flow regulation
BOW RIVER SUB-BASIN					
Cascade	L. Minnewanka	Cascade	1942	385[1]	Hydro
Spray Three Sisters Rundle	Spray Lakes	Spray	1951	422[1]	Hydro
Interlakes	Upper Kananaskis L.	Kananaskis	1942	105[3]	Hydro
Pocaterra	Lower Kananaskis L.	Kananaskis	1955	63[1]	Hydro
Barrier	Barrier	Kananaskis	1947	26	Hydro
Kananaskis	-	Kananaskis	1913	"Pondage"[1]	Hydro
Horseshoe	Horseshoe	Bow	1911	"Pondage"[1]	Hydro
Ghost	Ghost	Bow+Ghost	1929	132[1]	Hydro
Bearspaw	Bearspaw	Bow	1954	"Pondage"	Municipal/Hydro
Glenmore	Glenmore	Elbow	1932	28[1]	Municipal
Little Bow Canal	-	Highwood	1912	-	Irrigation
Squaw Coulee Diversion	-	Highwood	1956	-	Irrigation
Bassano	Newell	Bow	1910	319[3]	Irrigation
OLDMAN RIVER SUB-BASIN					
Waterton	Waterton	Waterton	1964	173[1]	Irrigation
St. Mary	St. Mary	St. Mary	1952	395[1]	Irrigation
Belly R. Diversion	-	Belly	1956	-	Irrigation
Murray	Murray L.	Seven Persons Cr.	1921	31[1]	Irrigation
Travers	Travers	Little Bow	1953	326	Irrigation
SOUTH SASKATCHEWAN RIVER SUB-BASIN					
Gardiner	L. Diefenbaker	S. Saskatchewan	1967	9 868	Multi-purpose
Duncairn	Reid L.	Swift Current Cr.	1942	103	Water supply
SASKATCHEWAN RIVER SUB-BASIN					
Squaw Rapids	Tobin L.	Saskatchewan	1962	2 195[1]	Hydro

Sources

1. Register of Dams in Canada
2. Alberta Research Council Hydrology Report 70-1
3. Prairie Provinces Water Board, Water Demand Study 1982.

Station	Name	Area* sq.km	Sediment yield t/km²/yr
05AB029	Meadow Ck	130	69
05AD002	Belly R.	1135	186
05AD028	Waterton R.	344	92
05AE002	Lea Creek	307	222
05EE003	Vermilion R.	613	4
05EE004	Vermilion R.	1573	9
05EE007	Vermilion R.	1106	1
05FE004	Battle R.	6620	14
05FF001	Battle R.	9394	133
05GC006	Eagle Creek	2094	4
05KB005	Burntout Bk	285	39
05KB006	Leather R.	122	6
05KB011	Doghide R.	434	39
05AB030	Streeter Ck	6	5
05CA003	Deer Creek	5	3

* sediment contributing area

TABLE 2.1

MEAN SUSPENDED SEDIMENT YIELDS

AT WRB MISCELLANEOUS STATIONS IN SASKATCHEWAN RIVER BASIN

(data from Ashmore, 1986)

Station	Name	Record	Concentration mg/L
05AB029	Meadow Ck	1972	6 110
05AD002	Belly R.	1974-83	14 700
05AD028	Waterton R.	1974-83	1 120
05AE002	Lea Creek	1972	1 820
05EE003	Vermilion R.	1974-83	68
05EE004	Vermilion R.	1976-79	173
05EE007	Vermilion R.	1981-83	233
05KC002	Connell Ck	1977-81	875
05FE004	Battle R.	1980-83	1 260
05FF001	Battle R.	1979-83	1 430
05GC006	Eagle Creek	1975-80	1 130
05KB005	Burntout Bk	1975-84	652
05KB006	Leather R.	1972-79	700
05KB011	Doghide R.	1977-83	786
05AB030	Streeter Ck	1966-69	432
05CA003	Deer Creek	1968-69	52

TABLE 3.1

MAXIMUM SUSPENDED SEDIMENT CONCENTRATIONS

AT WRB MISCELLANEOUS STATIONS IN SASKATCHEWAN RIVER BASIN

(data from Ashmore, 1986)

Year	Treatment	Runoff volume		Sediment Conc.‡		Sediment Loss§
		Rep. I	Rep. II	Rep. I	Rep. II	
		— mm —		— mg/liter —		kg/ha
1970-1971	Stubble	53	72	200	300	168
	Summerfallow	98	–	1,900	–	1,862
	Fall-fertilized summerfallow¶	49	–	3,500	–	1,715
1971-1972	Stubble	11	3	100	–	7
	Summerfallow	19	–	500	–	95
	Fall-fertilized summerfallow	7	–	700	–	49
1972-1973	Stubble	8	14	34	109	7.5
	Summerfallow	1	–	131	–	1.4
1973-1974	Stubble	90	19	0.8	0.0	0.2
	Summerfallow	55	60	0.1	2.1	0.6
1974-1975	Stubble	40	68	0.2	0.7	0.2
	Summerfallow	34	14	2.7	0.4	0.4
1975-1976	Stubble	101	68	723	458	498
	Summerfallow	30	23	835	4,752	638
Average and standard deviation	Stubble	46 ± 33		169 ± 215		112
	Summerfallow	37 ± 12		903 ± 1,484		433
	Fall-fertilized summerfallow	28 ± 21		2,100 ± 1,400		882

TABLE 3.2

RUNOFF VOLUMES AND SEDIMENT CONCENTRATIONS ASSOCIATED WITH SNOWMELT ON FIELDS IN SASKATCHEWAN (from Nicholaichuk and Read, 1978)

Drain	NO3-N mg/L	TP in mg/L	TP filtered mg/L	Sediment in mg/L	PP in mgP/kg
A Vauxhall	0.33	0.194	0.066	90.8	1410
D Vauxhall	0.28	0.104	0.047	37.8	1508
E Vauxhall	0.29	0.071	0.048	28.3	813
F Vauxhall	0.25	0.315	0.066	300.6	328
A Hays	0.27	0.233	0.109	84.1	1474
TA Hays	0.28	0.074	0.039	22.4	1562
K Vauxhall	0.14	0.104	0.050	51.0	1059
Ronalane Wasteway	0.18	0.054	0.041	34.8	374

TABLE 5.1

NUTRIENT AND SEDIMENT CONCENTRATIONS IN IRRIGATION RETURN FLOWS, ALBERTA

(from Oosterveld and Carefoot, 1979)

Nutrient concentrations in runoff from a flood-irrigated farm and downstream concentrations

	Total-P (mg/liter)		Ortho-P (mg/liter)		NO_3-N (mg/liter)		Suspended sediment (mg/liter)	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD
1975								
Supply water	0.058		0.028		0.1			
Surface runoff crop-alfalfa	0.177	0.085	0.111	0.026	1.4	0.76		
Drain†	0.153	0.077	0.069	0.062	0.1	0.15		
1976								
Supply water	0.014		Trace		0.1		11.2	
Surface runoff crop-wheat	0.185	0.044	0.081	0.017	1.6	0.33	86.0	29
Drain†	0.067		0.022		0.1		38.0	
Drain (1976 mean)‡	0.058	0.038	0.026	0.025	0.2	0.13	14.0	9

†Monitoring location was 2.5 km downstream from surface runoff sampling site.
‡Average of samples taken at 2-wk intervals during the season.

TABLE 5.2 (from Oosterveld and McMullin, 1979)

Brown Chernozem

	Native		Cultivated		
	Upper	Mid	Upper	Mid	
Ah	666	721	541	540	Ap
Bmk	729	690	491	467	Bm
Cca	672	847			
Ck	649	883	578	596	Ck

Gray Luvisol

	Native			Cultivated			
	Upper	Mid	Lower	Upper	Mid	Lower	
L-H	1061	1146	1175				
Ae	204	218	293	427	304	340	Ap
Bt	258	288	339	353	388	415	Bt
Ck	595	567		551	569	619	Ck

TABLE 5.3

TOTAL PHOSPHORUS CONCENTRATIONS (mg/kg) IN SASKATCHEWAN SOILS BY SOIL TYPE, SLOPE POSITION AND SOIL HORIZON

(from Schoenau et al., 1989)

	Sand	Coarse silt	Fine silt	Coarse clay	Fine clay
Surface	290	631	850	1010	625
	-335	-780	-1504	-2005	1651
Sub-surface	181	422	783	900	535
	-228	-648	-1080	-1294	-891

TABLE 5.4

RANGE IN TOTAL PHOSPHORUS CONTENT OF SASKATCHEWAN SOILS (mg/kg) BROKEN DOWN BY GRAIN SIZE FRACTION

Estimated Heavy Metal and Phosphorus Loads at the Mouth of the Bow and Oldman Rivers, May–October, 1980 (metric tonnes)

Site	Parameter	Total load in water			Load on suspended sediment			Storage in biomass*
		High water	Low water	Total	High water	Low water	Total	Total
Bow River at BRR	Pb	10.3†	5.2†	15.5†	4.8	0.2	5.1	42.6
	Zn	5.2	1.4	6.6	19.1	0.3	19.5	338.0
	Cu	7.6	0.8	8.5	5.7	0.1	5.8	30.8
	Ni	20.6†	2.6†	23.2†	4.9	0.1	5.0	46.9
	Cr	9.9	1.3†	11.2†	4.6	0.1	4.7	23.5
	Cd	1.0†	0.5†	1.5†	0.1†	0.0†	0.1	2.1
	Co	5.1†	2.6†	7.7†	1.6	0.6	2.2	69.5
	Mn	107.4	4.6	112.1	104.5	3.7	108.2	1 522.0
	Fe	5 584.9	156.4	5 741.3	4 305.1	78.3	4 383.5	872.0
	Hg	8.9×10^{-3}	9.7×10^{-4}	9.9×10^{-3}	1.2×10^{-2}	1.6×10^{-4}	1.3×10^{-2}	0.1
	P	247.9	32.4	280.4	223.1	52.6	275.8	13 581.0
Oldman River at ORBI	Pb	15.9†	2.6†	21.2†	10.6	0.0	10.6	
	Zn	81.9	0.6	82.6	70.7	0.1	70.9	
	Cu	22.1	0.2	22.4	21.6	1.4	23.1	
	Ni	26.2	1.3†	27.5†	18.4	0.0	18.5	
	Cr	26.4	0.6†	27.1†	18.8	0.0	18.8	
	Cd	1.1†	0.2†	1.4†	0.3	0.0	0.3	
	Co	11.6†	1.3†	12.9†	6.7	0.0	6.7	
	Mn	429.2	2.4	431.6†	364.2	28.2	392.5	
	Fe	20 623.0	98.4	20 820.0	16 078.0	478.5	16 556.0	
	Hg	2.3×10^{-3}	1.9×10^{-3}	2.5×10^{-3}	3.3×10^{-2}	5.8×10^{-1}	6.2×10^{-1}	
	P	345.7	6.8†	352.6†	622.4	10.8	633.2	

*Assumes 100% plant cover; Alberta Environment suggests this should be reduced by at least 25%.
†Element not detectable; load assumes element has a concentration equal to detection limit; load is therefore maximized.

TABLE 6.1 HEAVY METAL LOADINGS IN BOW-OLDMAN SYSTEM
(from Blachford and Ongley, 1984)

Metal Concentrations in Bank Materials

Metal (mg/kg)	Site					
	ORPD*		ORBI	BRDB	BRBC*	
Cu	19	20	10	13	13	12
Pb	13	15	10	7	8	7
Zn	65	68	51	48	61	60
Co	7	7	5	7	6	6
Ni	19	23	18	19	19	20
Mn	218	209	262	266	327	338
Cr	25	22	22	27	21	19
Fe (%)	1.96	1.76	1.66	1.81	1.66	1.62
Cd	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Hg	0.061		0.040	0.027	0.040	

*Duplicate analyses reported.

ORPD: Oldman R., downstream from Lethbridge

ORBI: Oldman R., upstream from South Saskatchewan River

BRDB: Bow R., downstream of Bassano Dam

BRBC: Bow River at Bow City

TABLE 6.2

METAL CONCENTRATIONS IN BANK MATERIALS OF BOW AND OLDMAN RIVERS

(from Blachford and Ongley, 1984)

	Summer-fallow	A-horizon**			
	Horizon	Cultivated			Plant-available* (range)
	C	A		Non-cultivated	
Cd	1.0	0.9	0.9	1.0	0.07 - 0.40
Cr	16	23	19	27	
Cu	23	25	18	19	1.8 - 7.4
Hg	0.039	0.033	0.036	0.035	0.016- 0.041
Ni	39	42	34	39	1.2 - 8.1
Pb	19	17	16	16	5.2 - 7.4
Zn	66	116	100	119	6 -19

all values in ppm (mg/kg)

* Ap horizon (Dudas and Pawluk, 1977: Alberta)

** Mills and Zwarich, 1975: south and west Manitoba

TABLE 6.3

HEAVY METAL CONCENTRATIONS IN SURFACE SOILS OF PRAIRIES

	Reach	Approx. length of reach (km)	n	range	mean	95% limits
A	Devon-Edmonton	20	35	8-173	53	±13.6
B	Edmonton-Ft. Saskatchewan	20	6	20-157	75	±38.6
C	Ft. Saskatchewan-Redwater R.	29	4	80-124	106	±21.5
D	Redwater R.-provincial border	250	63	13-335	92	±16.0

n: sample size

range, mean, limits: all values in µg/kg

TABLE 6.4

LEVELS OF MERCURY IN BOTTOM SEDIMENTS OF NORTH SASKATCHEWAN RIVER, ALBERTA

(from AEC, 1983a)

	Saskatchewan River	Petaigan Bay
lead	15	25
mercury	0.084	0.046
chromium	89	84
zinc	77	91
nickel	11	36
copper	12	30
cobalt	11	9.5
arsenic	7.7	6.9
cadmium	0.2	0.4

all values in ug/g

TABLE 6.5

METAL CONCENTRATIONS IN BOTTOM SEDIMENTS OF TWO SITES IN TOBIN LAKE

(from Warwick, 1990, pers. comm.)

	Water solubility in ppm at 20-25°C	Vapour pressure at 20-25°C mm Hg x 10^{-6}	Approx. half-life in soil
Chlorinated hydrocarbon insecticides			
DDT	0.001-0.040	0.15	10-20 yrs
aldrin	0.01 -0.2	6	8 years
heptachlor	<1	300	2-4 yrs
methoxychlor	0.1 -0.2	XXX	
dieldrin	0.1 -0.25	0.18	8 years
endrin	0.23	0.2	8 years
toxaphene	0.4		
endosulfan	<1		
lindane	7 -10	9 - 45	2 years
Organophosphate insecticides			
parathion	24	38	<1 month
diazinon	40	140	1 month
malathion	145	40	
dimethoate	20000	100	
Carbamate insecticides			
carbaryl	40		2 weeks
carbofuran	700		2 weeks
Basic herbicide compounds			
atrazine	35	0.3	4 months
Acidic herbicide compounds*			
bromoxynil	130	XXX	
2,4,5-T	238	XXX	
MCPA	550	XXX	
2,4-D	650	XXX	
dicamba	4500	XXX	
picloram	>10%		
Cationic compounds			
paraquat	70%		> 1 year
diquat	70%		

XXX denotes non-volatile

* solubility varies with type of compound (salt, ester etc.)

TABLE 7.1 CHARACTERISTICS OF SELECTED PESTICIDES

[based on data complied by Weber (1972) and Wauchope (1978)]

Pesticide Group	Compounds Investigated*
Organochlorine	aldrin
	α-BHC
	γ-BHC (lindane)
	p, p'-DDD
	p, p'-DDE
	p, p'-DDT
	dieldrin
	α-Endosulfan
	β-Endosulfan
	endrin
	heptachlor
	heptachlor epoxide
	p, p'-methoxychlor
Chlorophenoxy acid	2,4-D
	2,4-DB
	2,4-DP
	2,4,5-T
	MCPA
Organophosphorus	dimethoate
	Guthion
	malathion
Carbamates	Baygon
	carbofuran

*Capitalized name indicates a brand name

TABLE 7.2

PESTICIDES AND METABOLITES INVESTIGATED 1971-77
IN SURFACE WATER OF PRAIRIES

(from Gummer, 1979)

Province	On-farm costs* in $ millions	Farmland area** in million ha	$ per ha
Ontario	68	6.0	11.33
Alberta	200	19.1	10.47
Atlantic	11	1.2	9.17
Saskatchewan	220	25.9	8.49
B.C.	10	2.2	4.55
Quebec	10	3.8	2.63
Manitoba	10	7.6	1.32

* from Rennie (1985)

** from Statistics Canada (1981)

TABLE 8.1

ANNUAL ON-FARM COSTS OF SOIL EROSION
BY PROVINCE

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1.4 Soil organic matter losses from 'A' horizons in different soil zones of the Prairie Provinces
1.5 District irrigation area, Saskatchewan-Nelson catchment, by sub-basin, 1976
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5.10 Phosphorus speciation of suspended sediments in Blachford-Ongley study
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7.1 Pesticide levels in Prairie surface water, 1971-77
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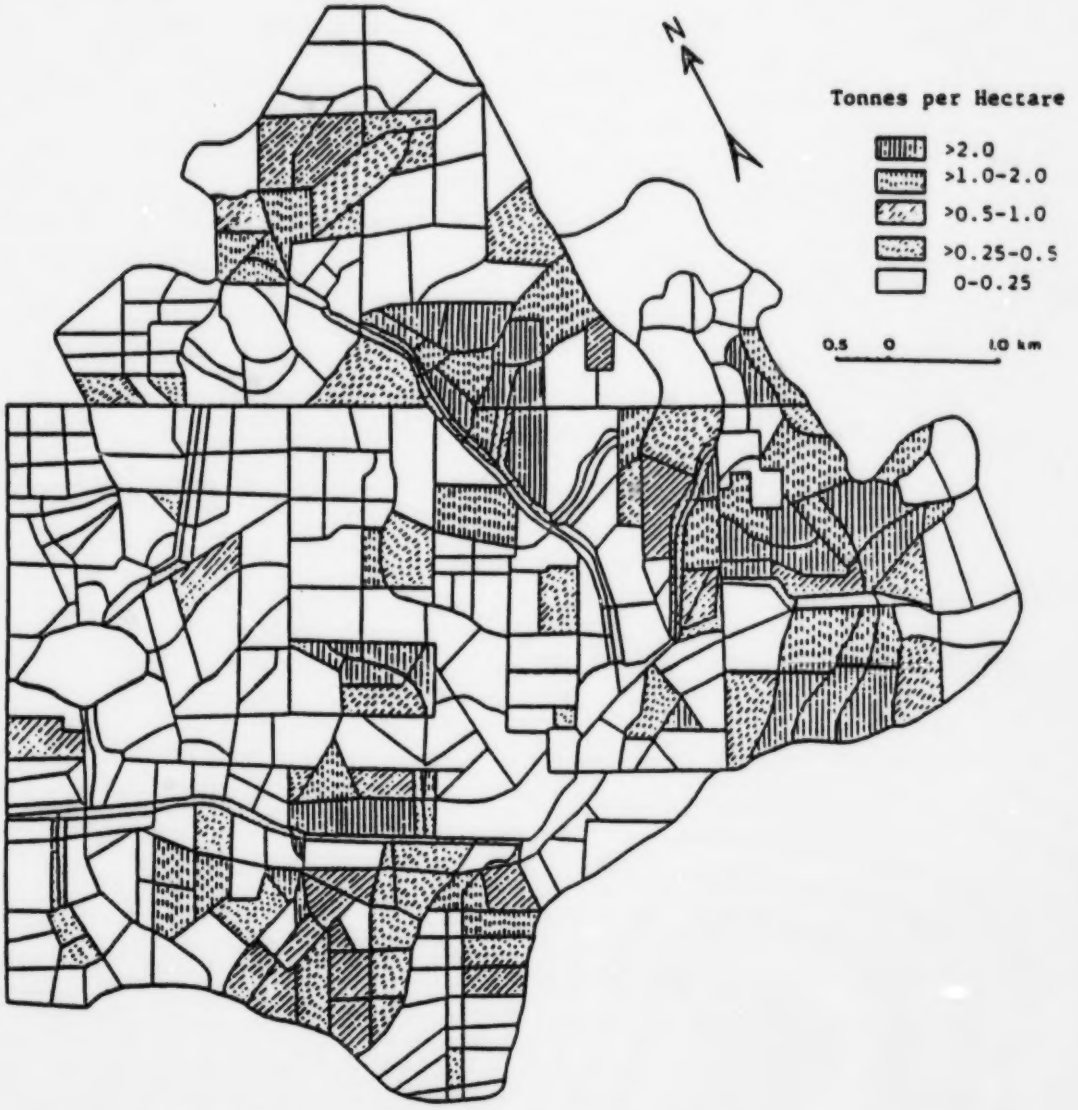
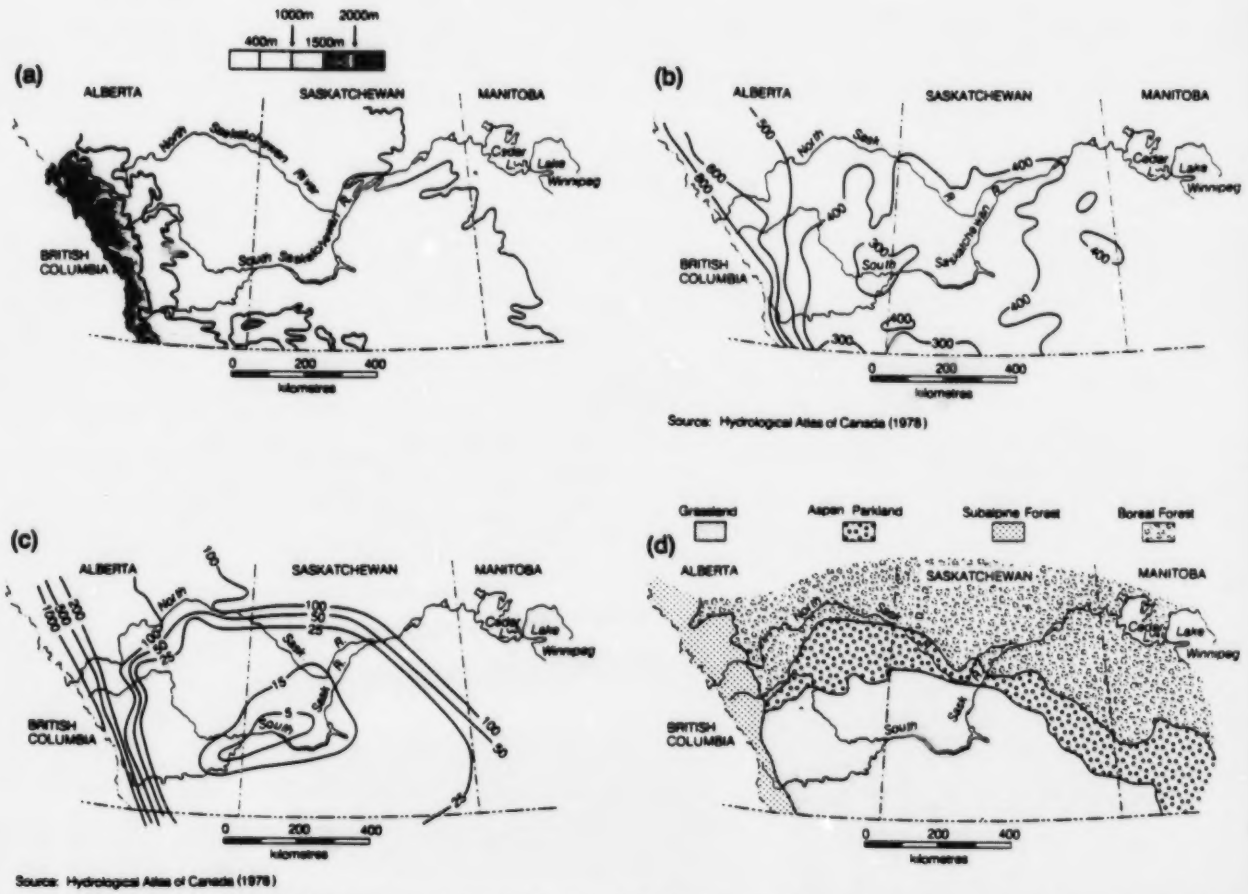
7.4 Organic and particle size composition of suspended sediment in the Bow and Oldman rivers by season



Spatial variation of spring sediment yield on Stratford–Avon Basin 1, as generated by GAMES (Dickinson and Pall 1982).

FIGURE 1.1

MAP OF SEDIMENT YIELDS IN A SMALL FARMED BASIN. ONTARIO
(from Dickinson and Pall. 1982)



Physical geography of the Saskatchewan River basin: (*a*) relief; (*b*) mean annual precipitation (mm); (*c*) mean annual runoff (mm); (*d*) natural vegetation.

FIGURE 1.2

PHYSICAL GEOGRAPHY OF THE SASKATCHEWAN RIVER BASIN
(from Ashmore and Day, 1988)

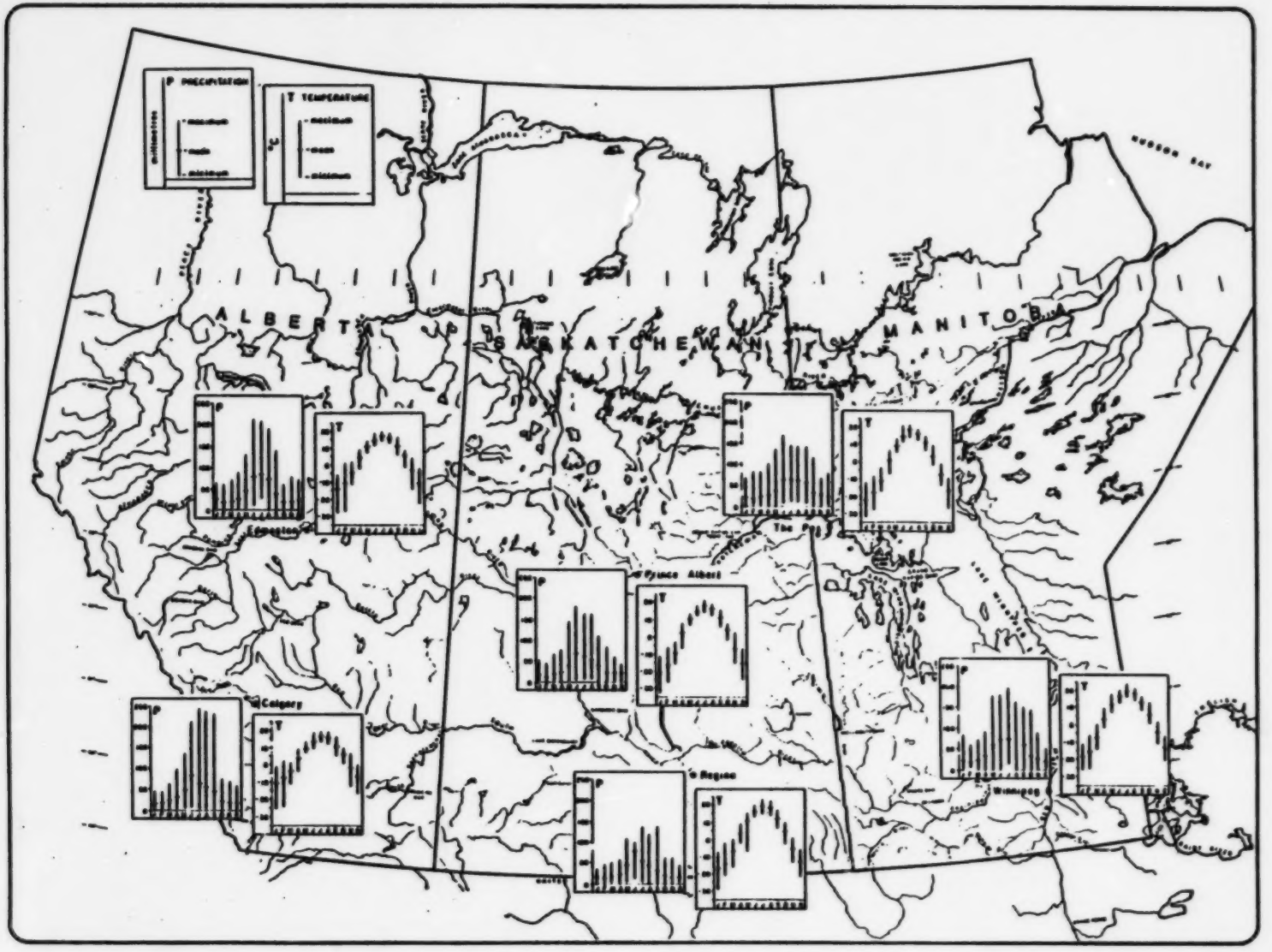


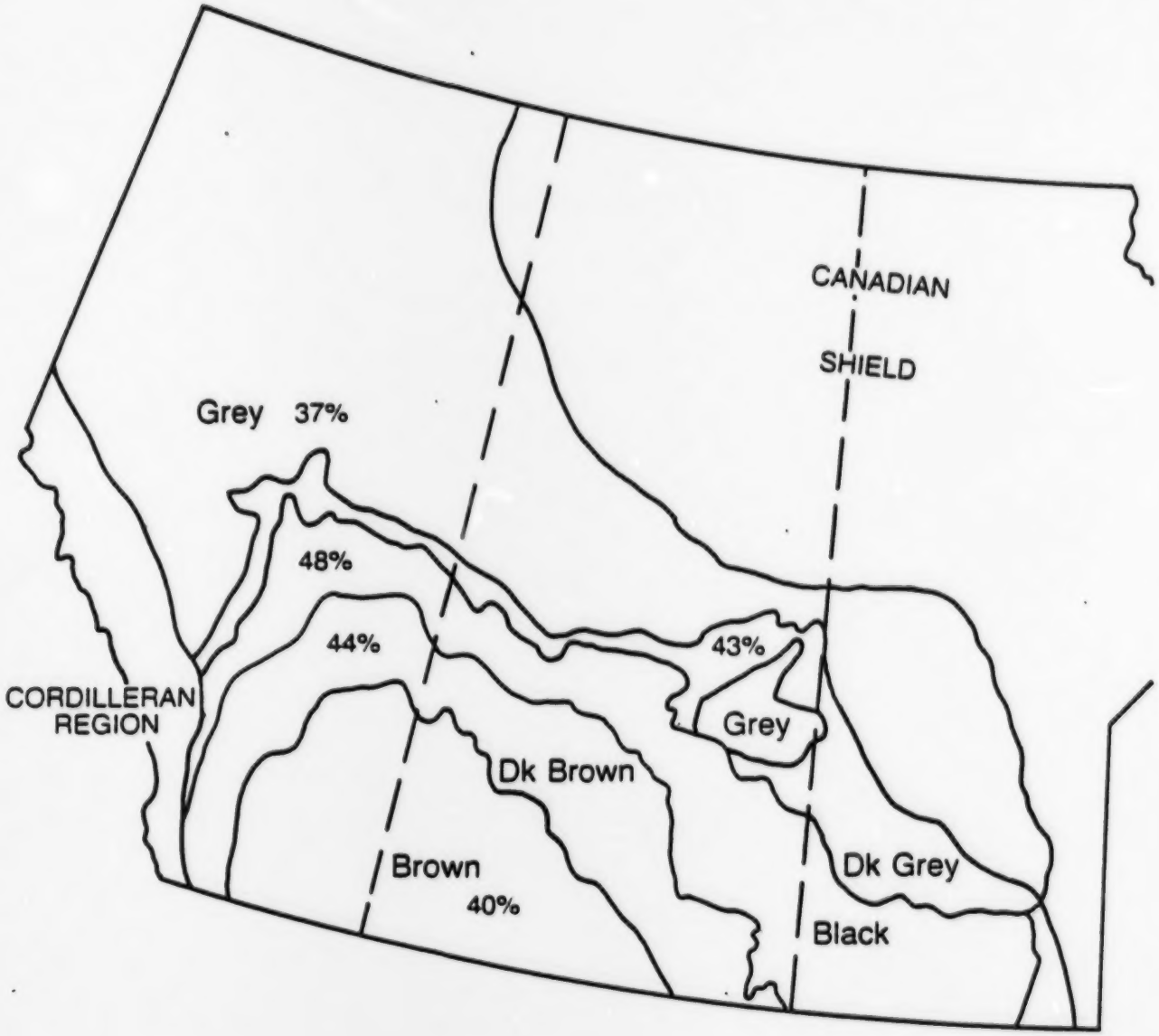
Fig. 1.3 **Temperature and Precipitation Patterns**

(from PPWB, 1982)

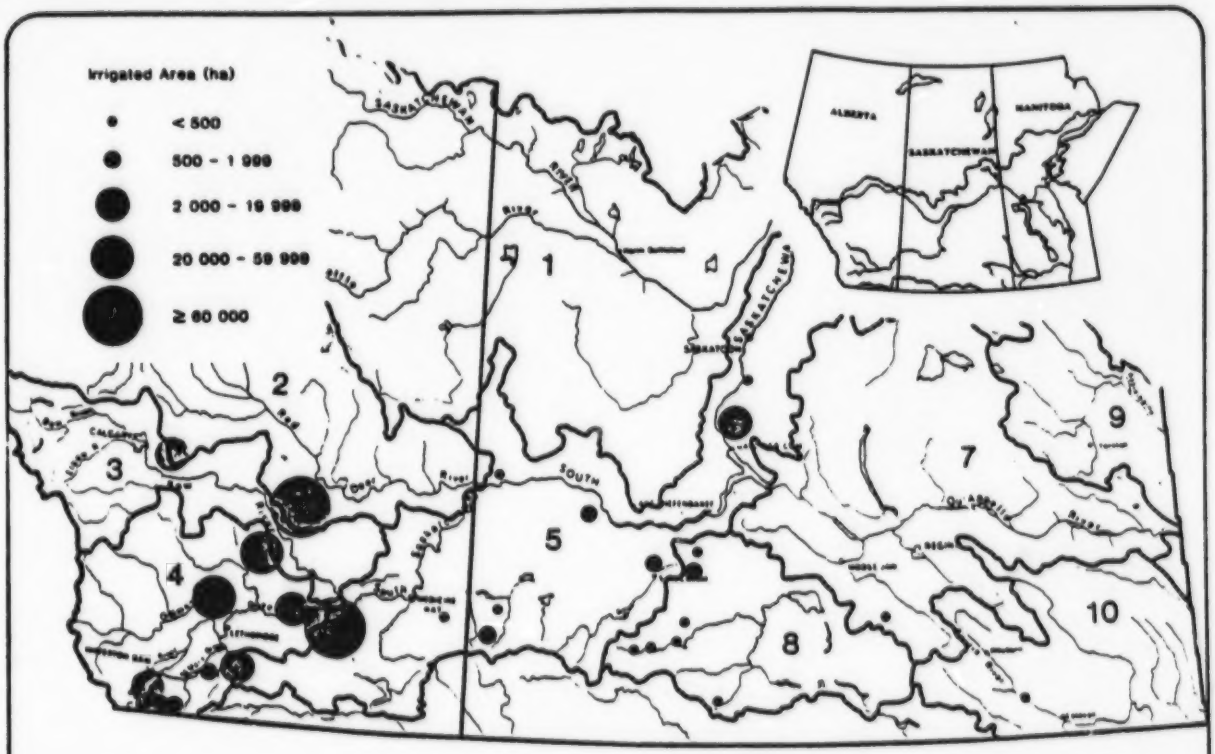
FIGURE 1.4

SOIL ORGANIC MATTER LOSSES FROM 'A' HORIZONS, BY SOIL ZONE, PRAIRIES

(from Anderson and Knapik, 1984)



SOURCE: Mc Gill, Campbell, Dormaar, Paul, and Anderson (1981,#48), p. 80.



SUB-BASIN NUMBER	SUB-BASIN		IRRIGABLE AREA*(ha)	IRRIGATED AREA (ha)	PERCENT IRRIGATED
2	RED DEER RIVER	(A)	61 791	54 878	89
3	BOW RIVER	(A)	77 556	69 217	89
4	OLDMAN RIVER	(A)	166 857	136 707	82
5	SOUTH SASKATCHEWAN RIVER	(A)	71 591	59 325	83
		(S)	12 869	11 740	90
			84 460	71 065	84
7	QU'APPELLE RIVER	(S)	109	48	44
8	OLD WIVES LAKE	(S)	2 610	2 485	95
10	SOURIS RIVER	(S)	62	43	68
	SASKATCHEWAN-NELSON BASIN	(A)	377 797	320 127	85
		(S)	15 670	14 317	91
			393 467	334 444	85

*Defined to be land in an irrigation district that is assessed as "irrigable".

Figure 1.5 **District Irrigation Area by Sub-basin, 1976**

(from PPWB, 1982)

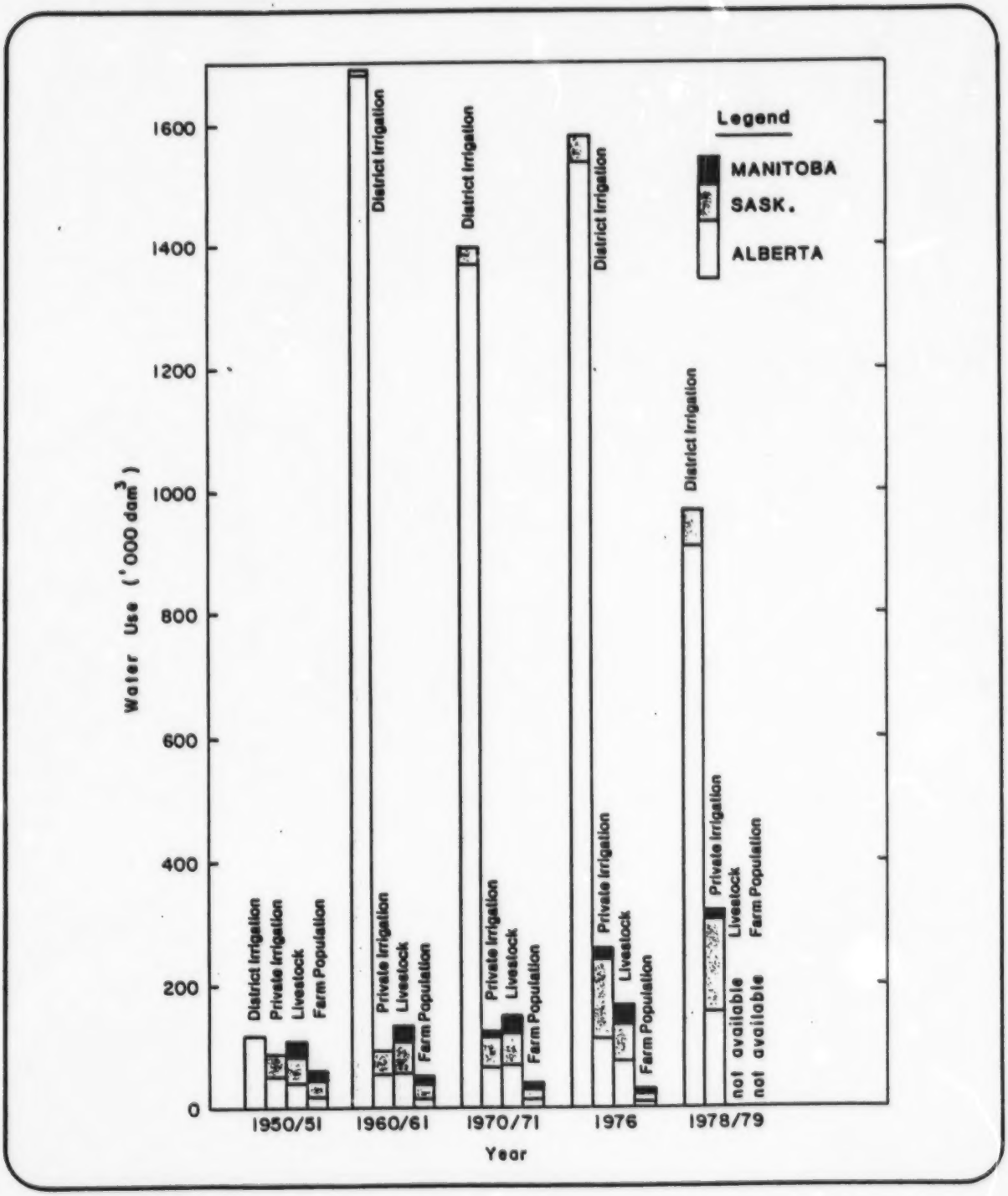


Figure 1.6 **Agricultural Water Uses**

(from PPWB, 1982)

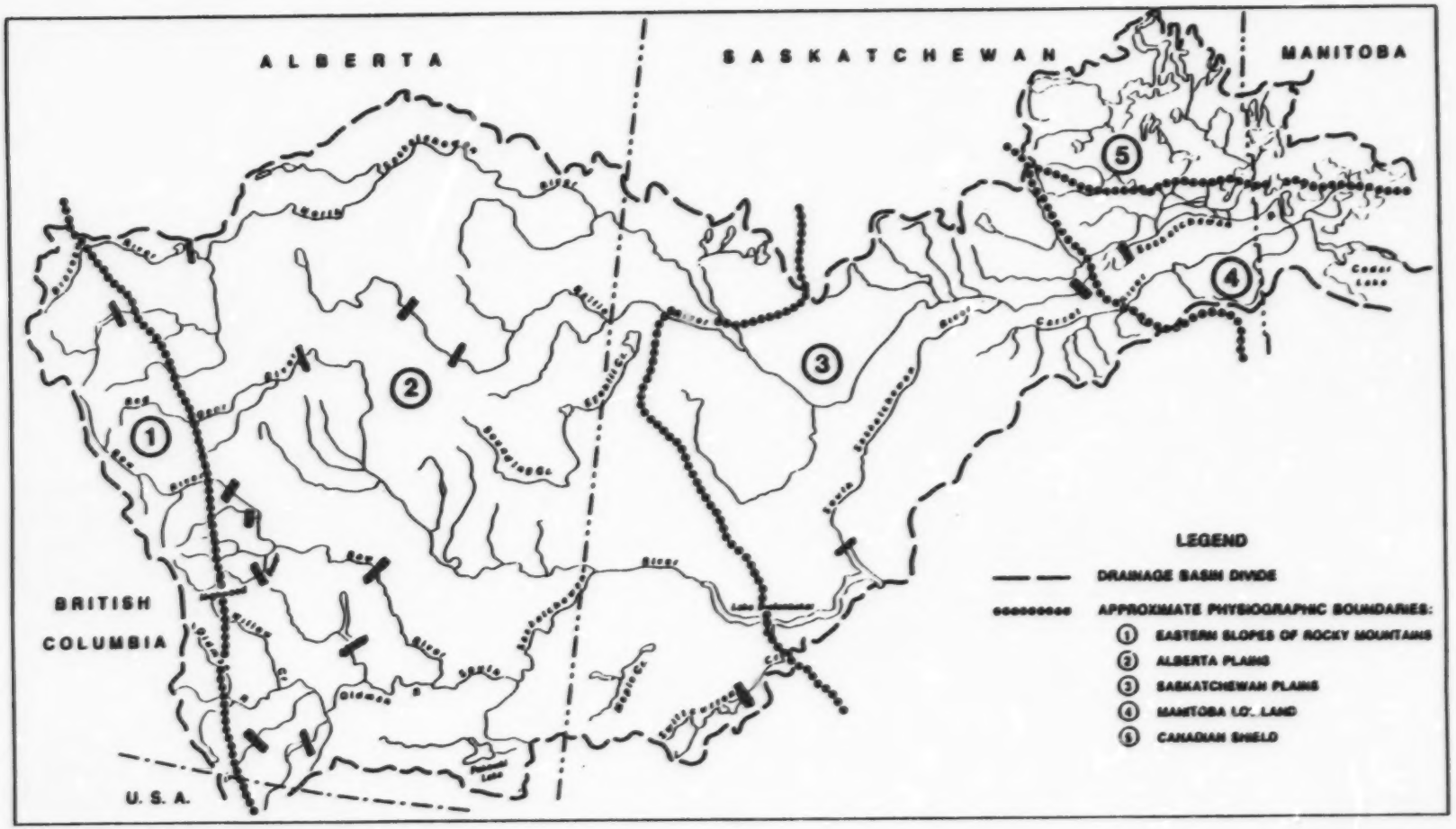


Figure 1.7 Drainage, physiography and location of major dams of the Saskatchewan River basin

(from Ashmore and Day, 1988)

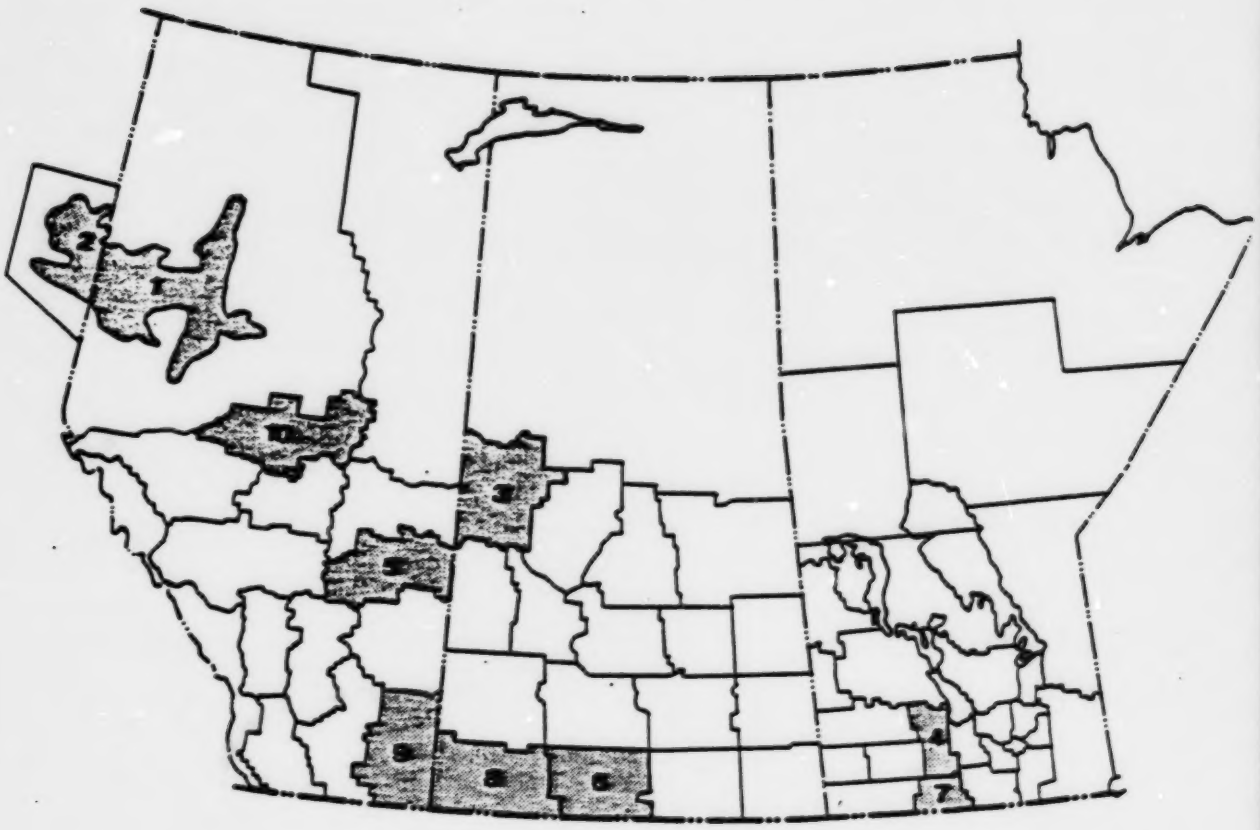
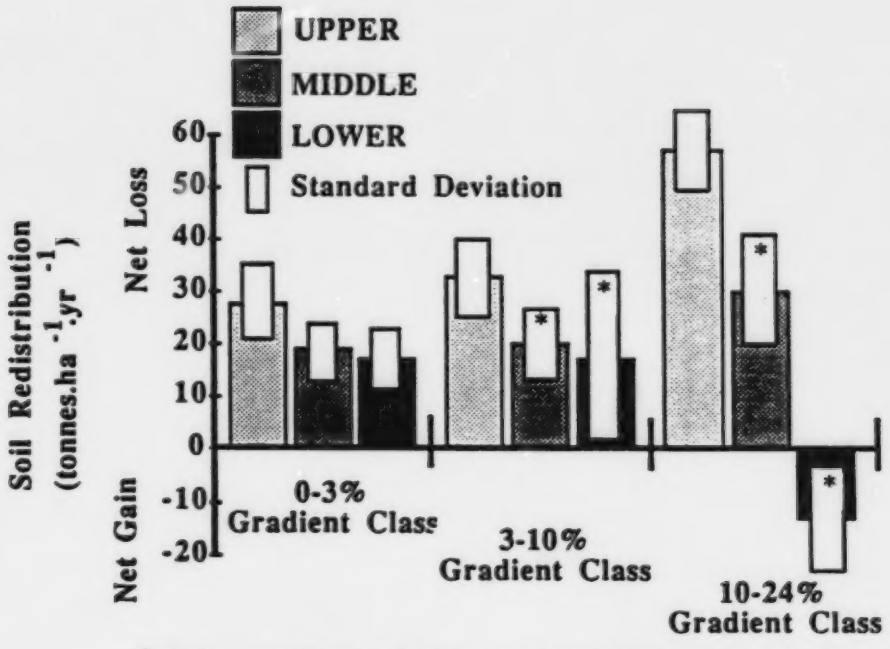


Figure 2.1

Location of census divisions most affected by erosion, ranked by annual cost per improved acre.

(from Anderson and Knapik, 1984)



* Both net soil losses and gains were observed in these slope positions.

Mean soil redistribution by gradient class and idealized hillslope position.

FIGURE 2.2 NET SOIL LOSSES AND GAINS AT SITES ON SLOPING FARMLAND, NEAR UNITY, SASKATCHEWAN

(from Kiss et al., 1986)

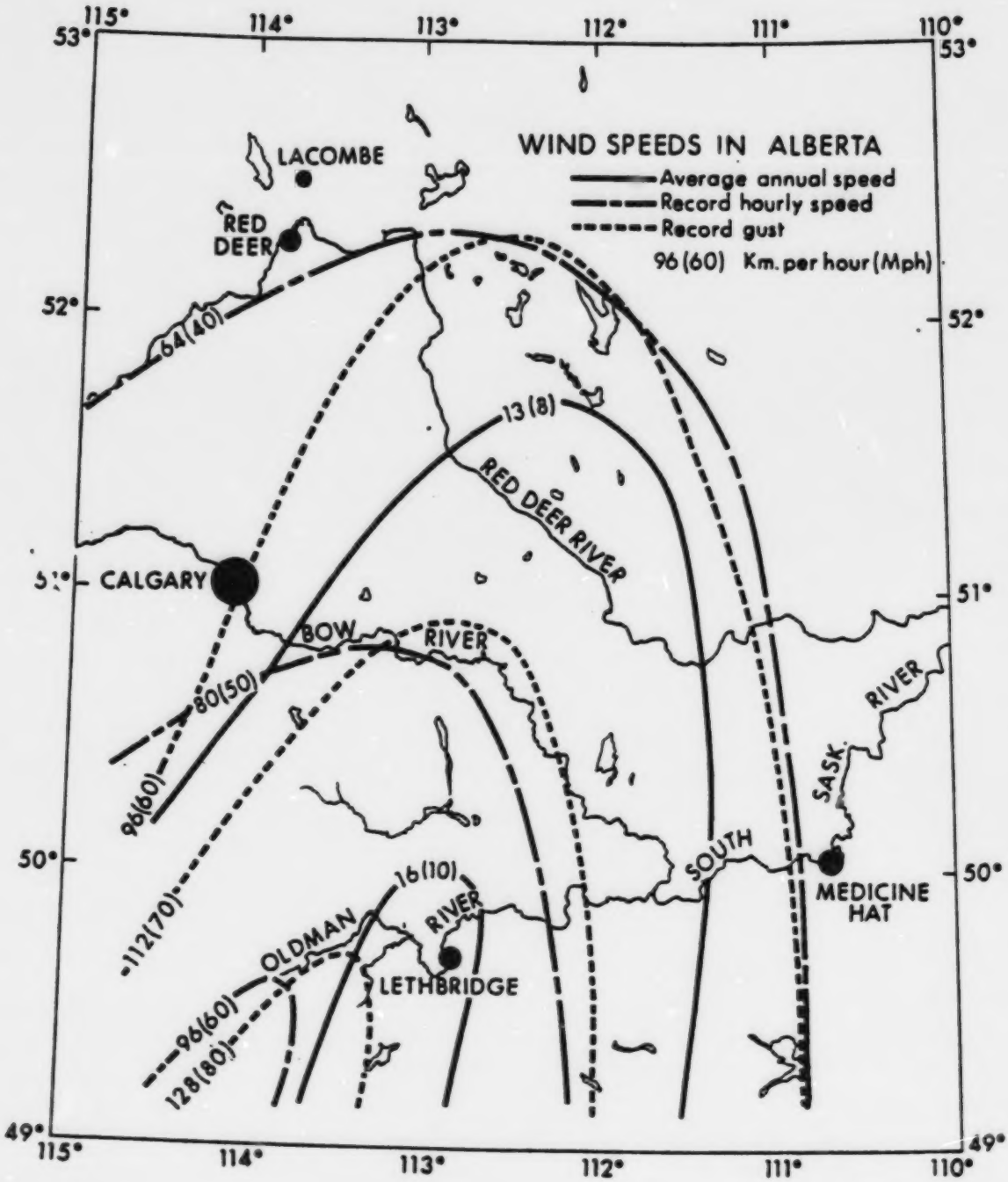


FIGURE 2.3 WIND SPEEDS IN SOUTH-EASTERN ALBERTA

(from Toogood, 1971)

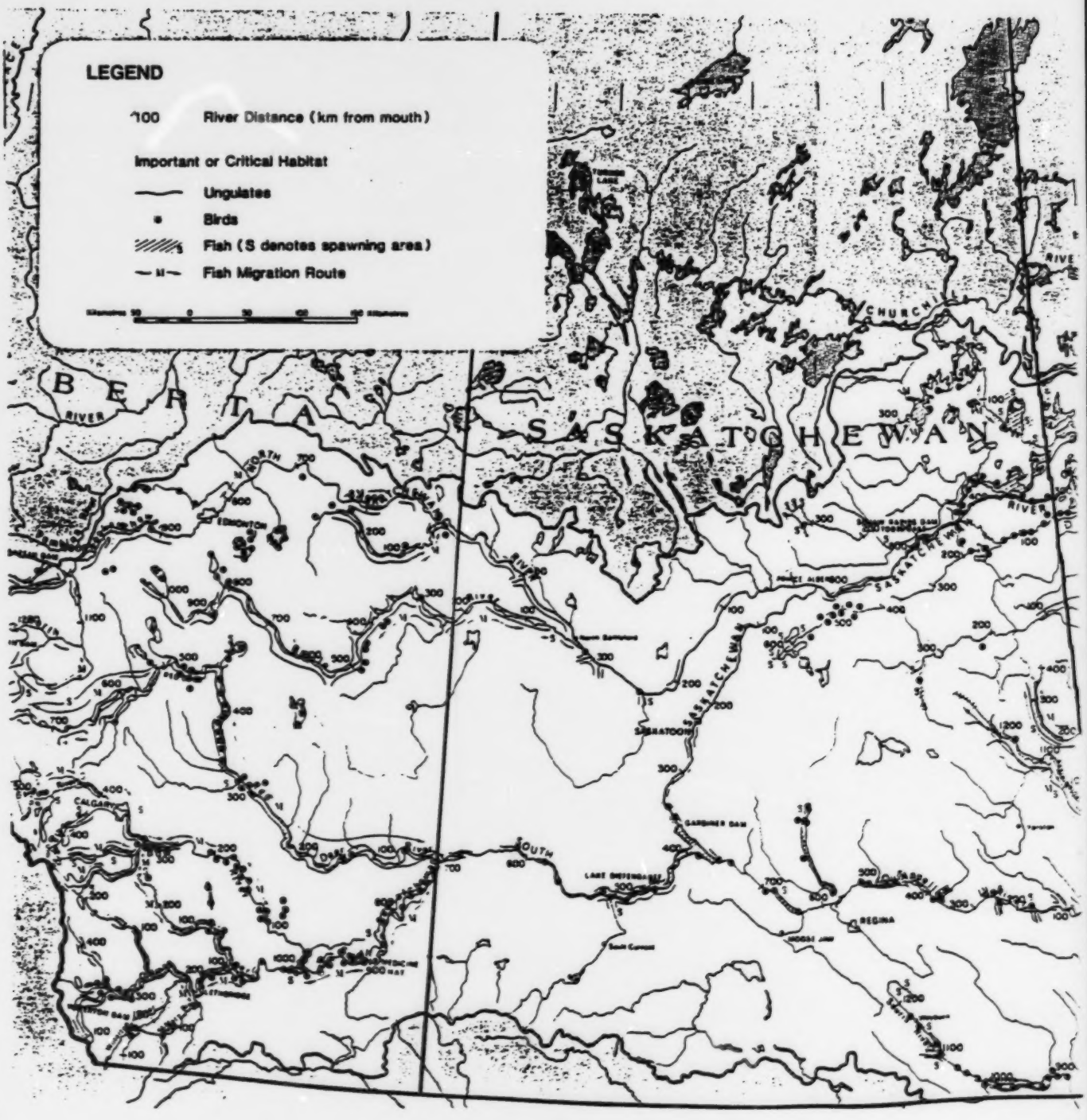
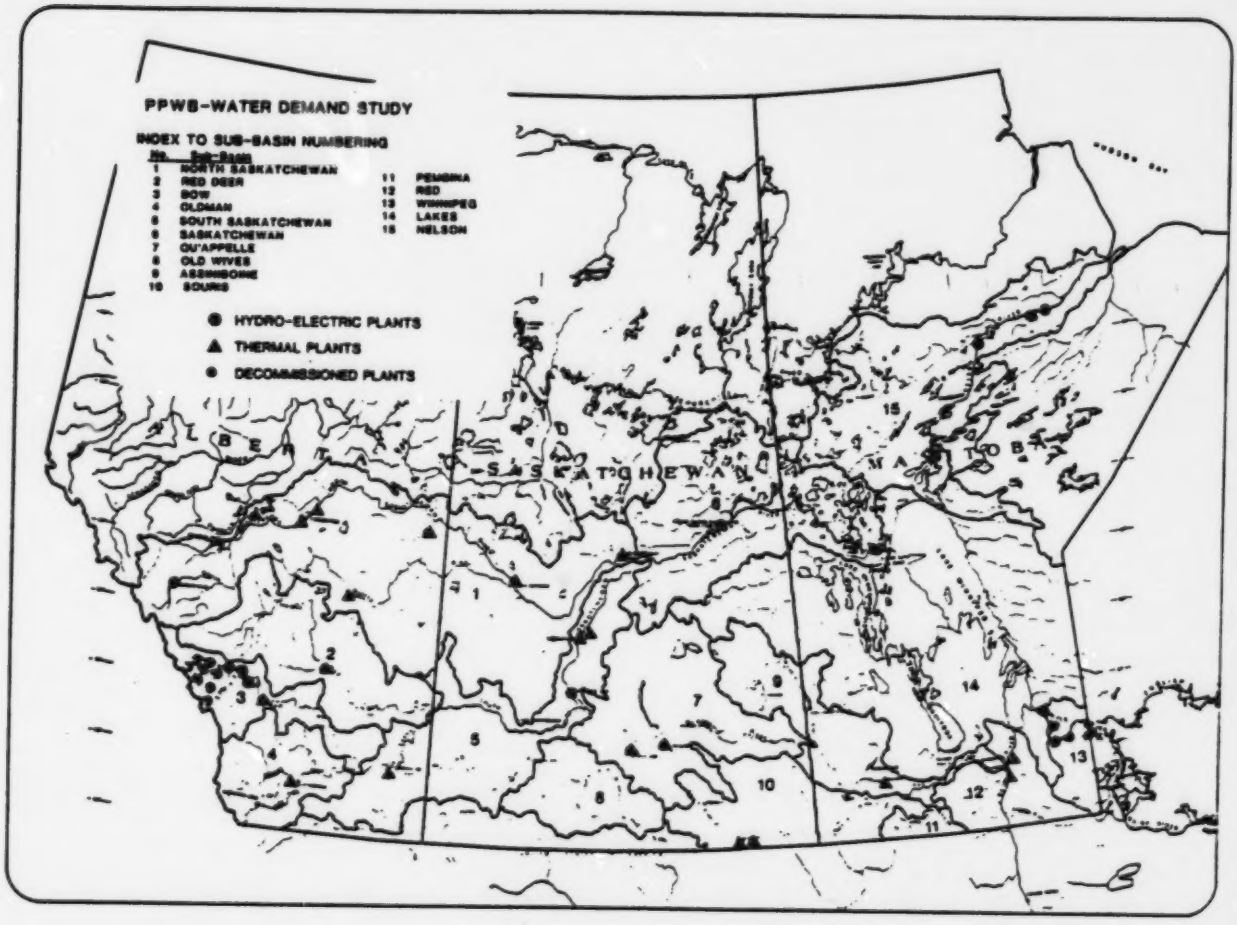


FIGURE 3.1

LOCATIONS OF IMPORTANT FISH AND WILDLIFE HABITAT IN THE ALBERTA AND SASKATCHEWAN PARTS OF THE PRAIRIE PROVINCES

(from PPWB. 1982)



Power Plants, 1951-78

FIGURE 3.2

LOCATIONS OF POWER PLANTS IN THE SASKATCHEWAN-NELSON RIVER BASIN. 1951-78 (from PPWB. 1982)

Drainage system, physiographic regions, and the location of long-term Water Survey of Canada sediment stations.

FIGURE 3.3

LOCATION OF LONG-TERM WRB SEDIMENT STATIONS IN THE SASKATCHEWAN RIVER BASIN (from Ashmore and Day, 1988)

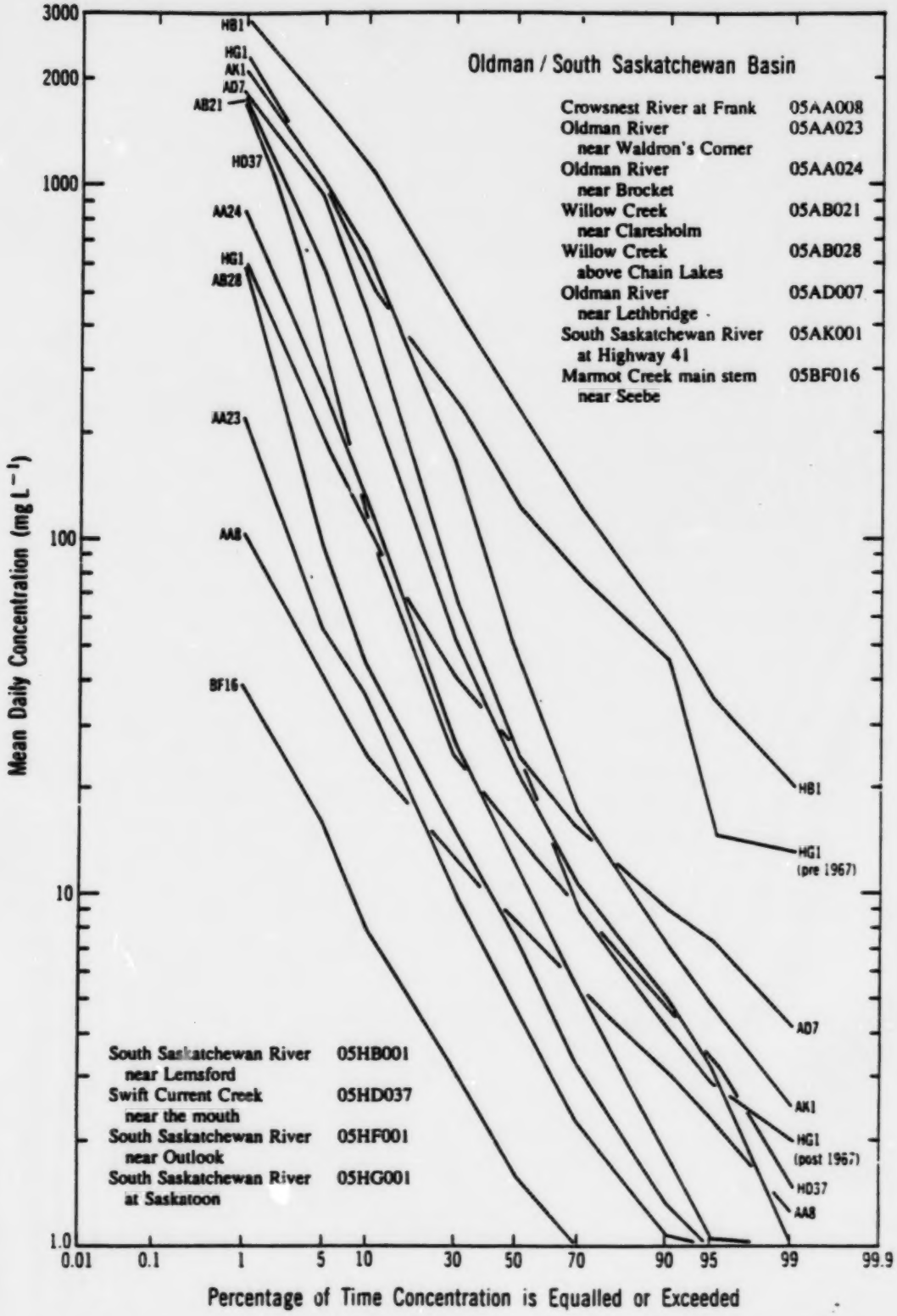


Figure 3.4 Seasonal Daily Concentration Duration Curves for Long-Term Stations

(from Ashmore, 1986)

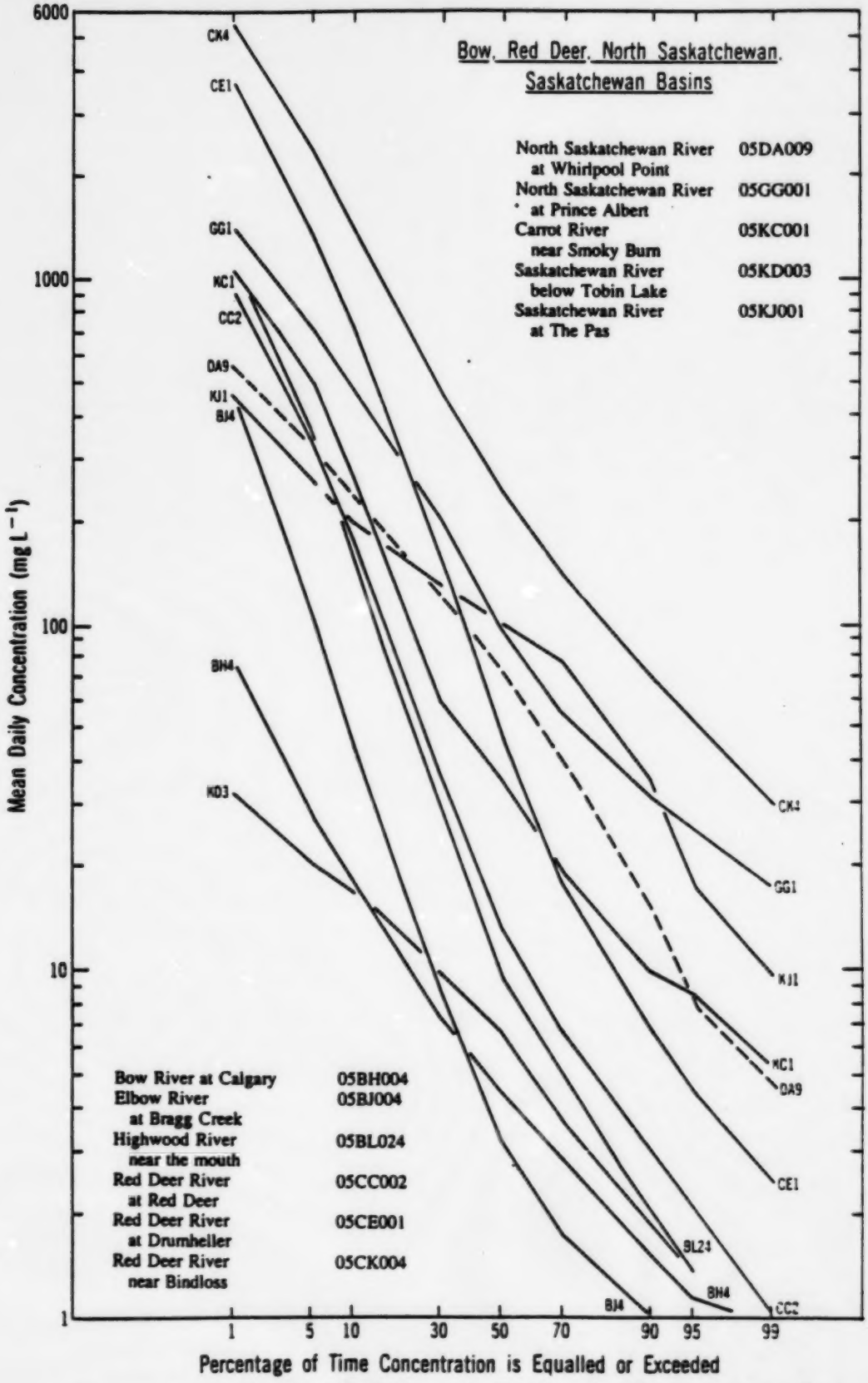


Figure 3.5 Seasonal Daily Concentration Duration Curves for Long-Term Stations (from Ashmore, 1986)

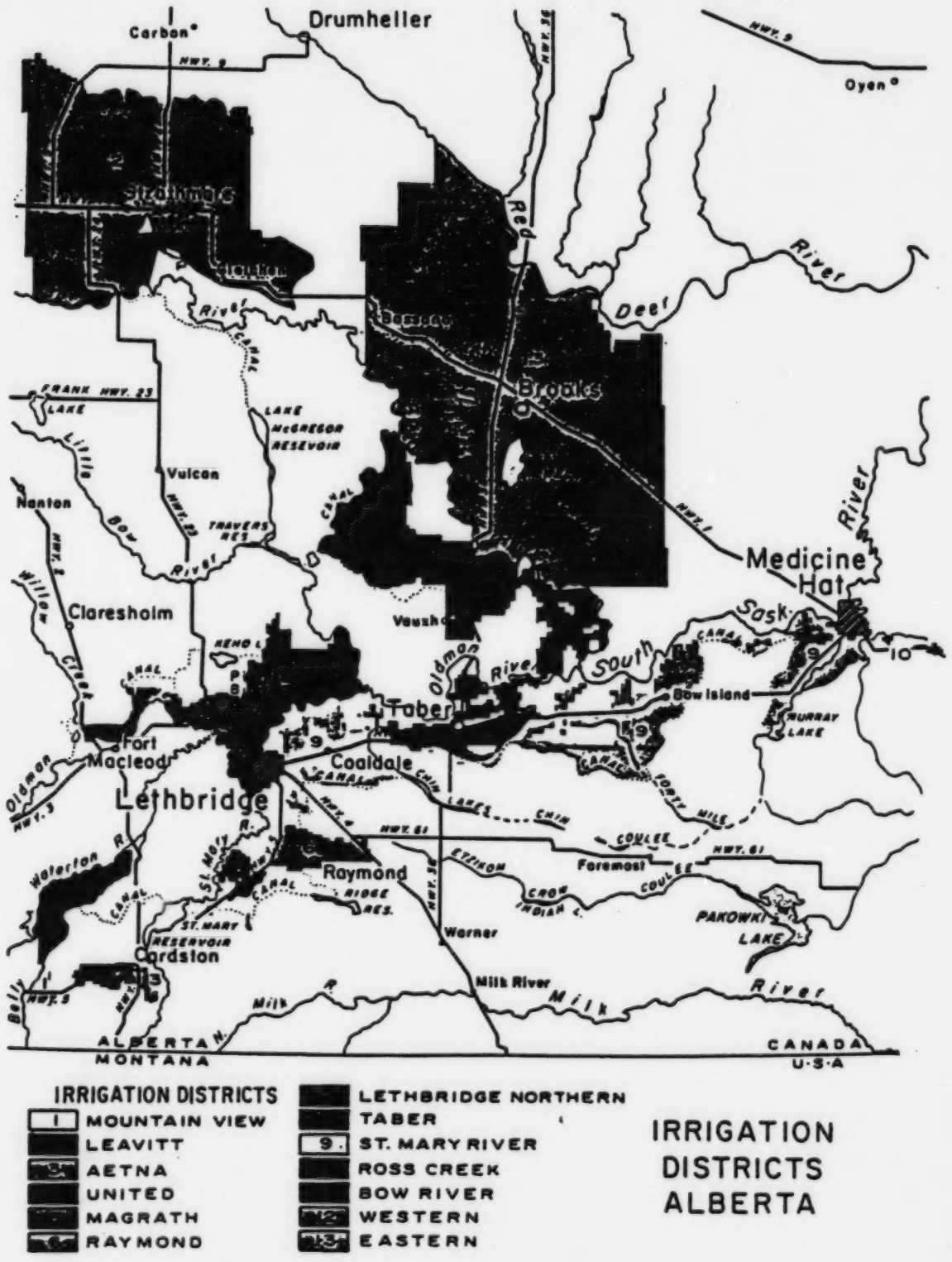


Figure 4.1 Irrigation Districts in Alberta (from PPWB, 1982)

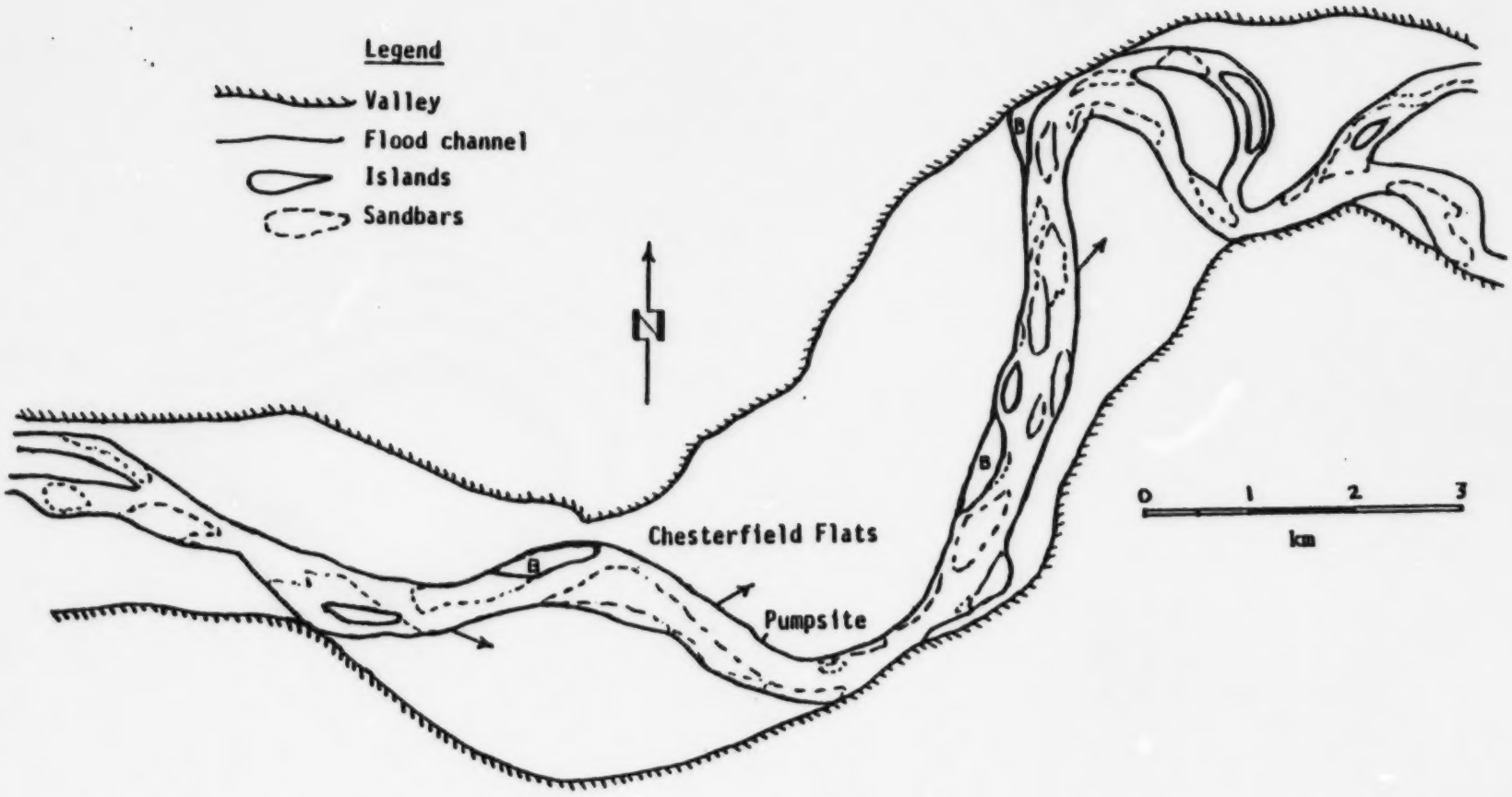


Figure 4.2 Map of South Saskatchewan River and valley at Chesterfield Flats (from Smith and Wigham, 1990)

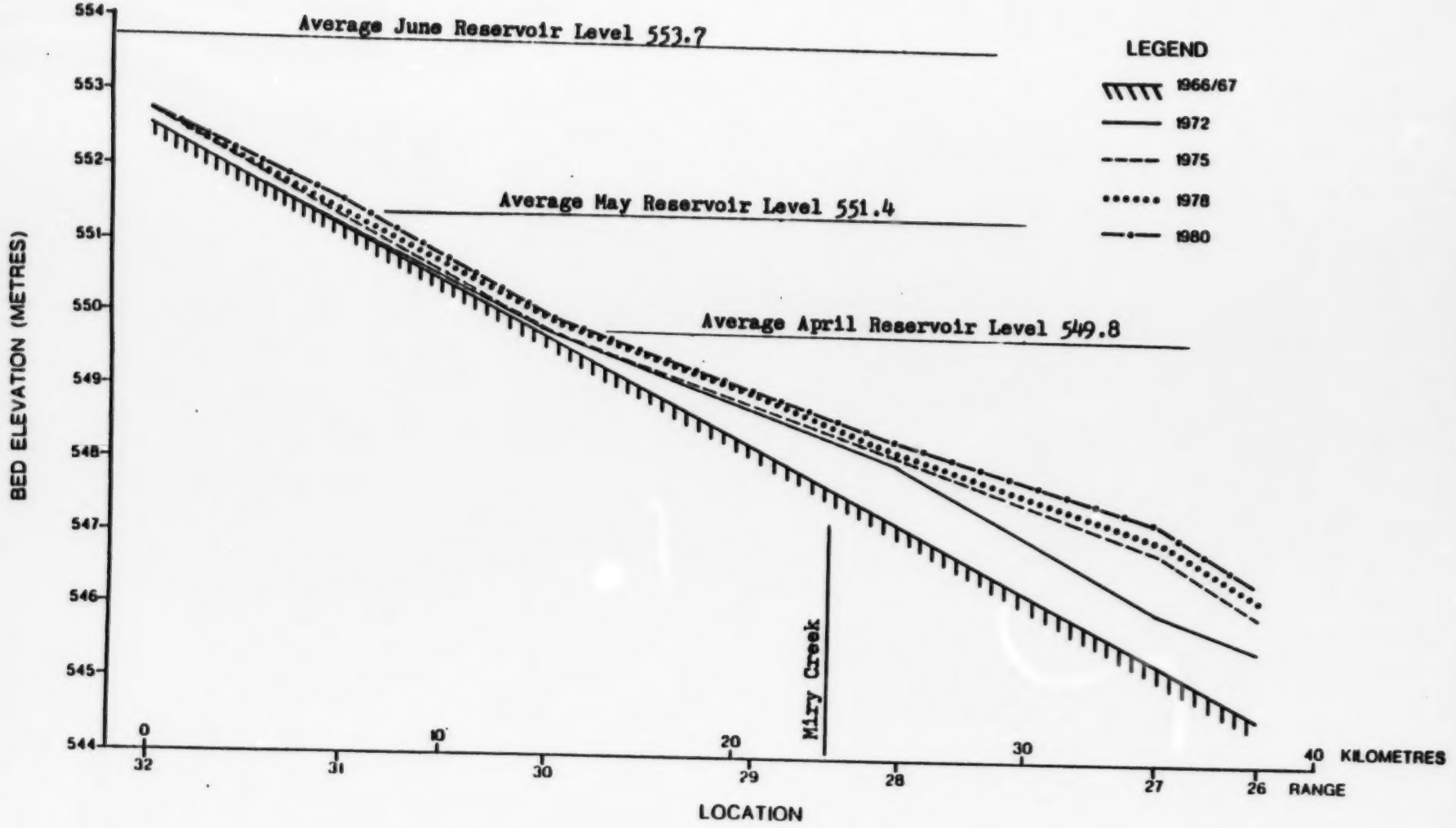


Figure 4.3 Bed Elevation Changes, South Saskatchewan River Range 32 to 26 (after Yuzyk, 1983)

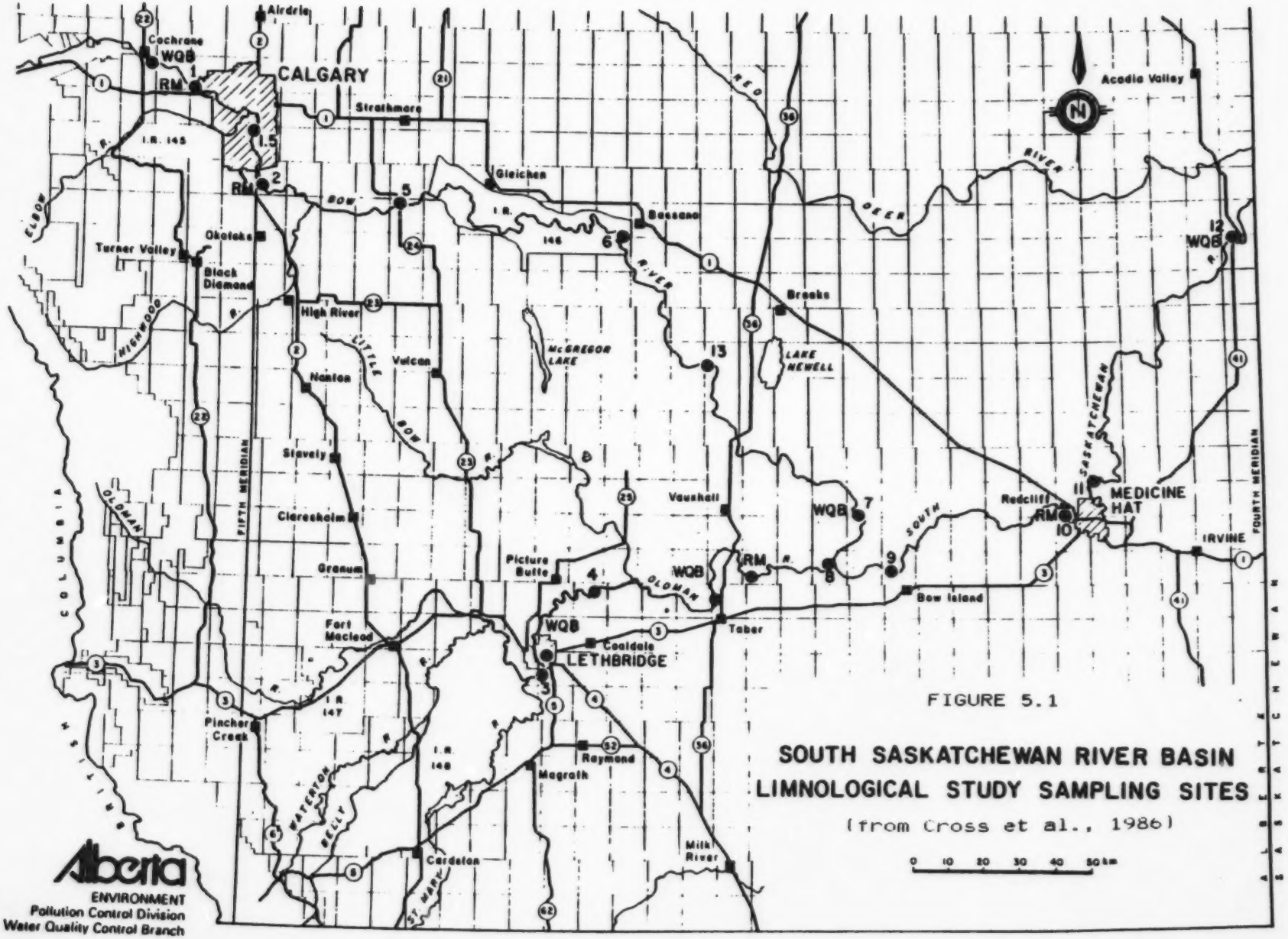


FIGURE 5.1

SOUTH SASKATCHEWAN RIVER BASIN
LIMNOLOGICAL STUDY SAMPLING SITES

(from Cross et al., 1986)

Alberta
ENVIRONMENT
Pollution Control Division
Water Quality Control Branch

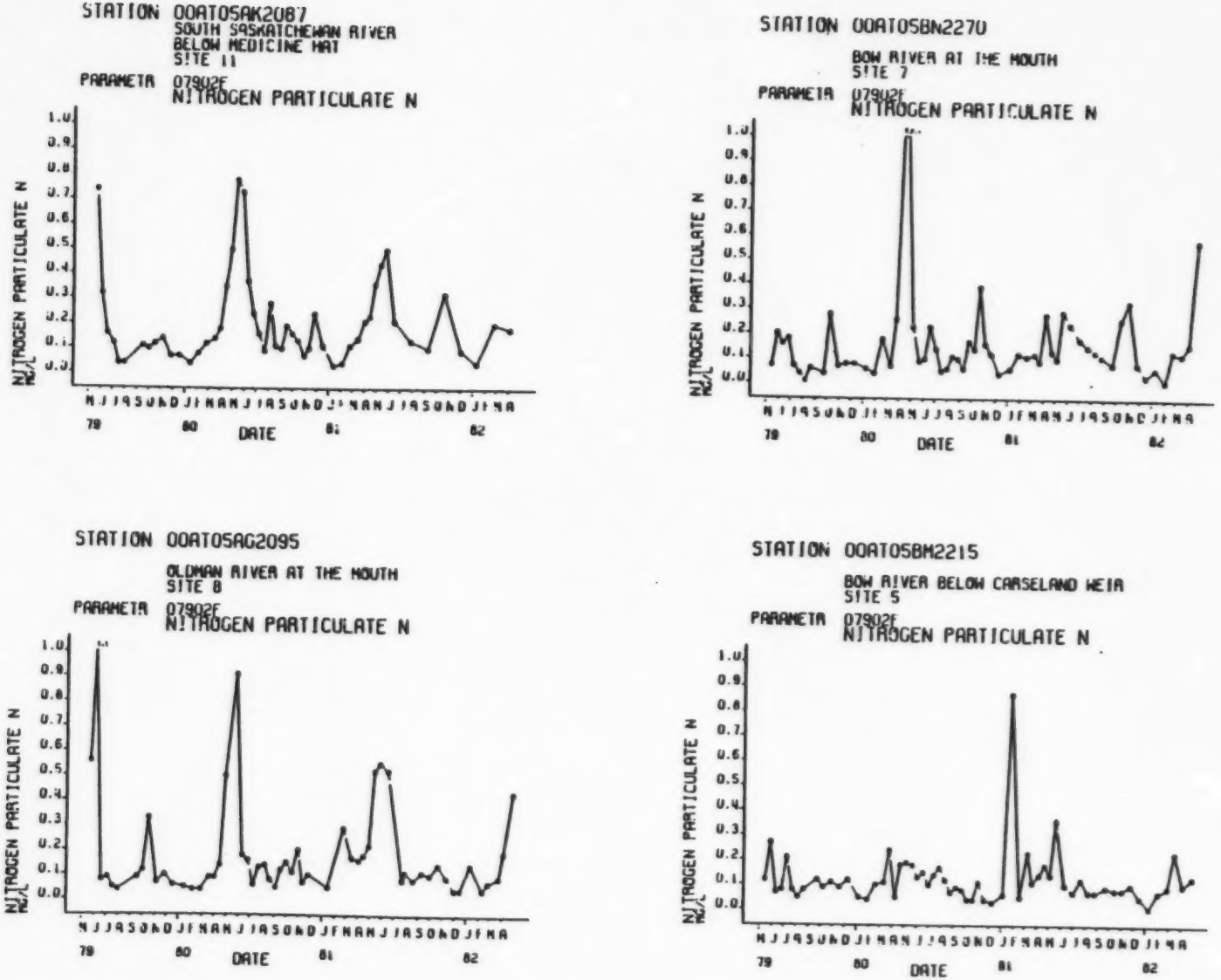


Fig. 5.2 Comparison of particulate nitrogen data from four sites in the study basin.

(from Cross et al., 1986)

Alberta
ENVIRONMENT
Pollution Control Division
Water Quality Control Branch

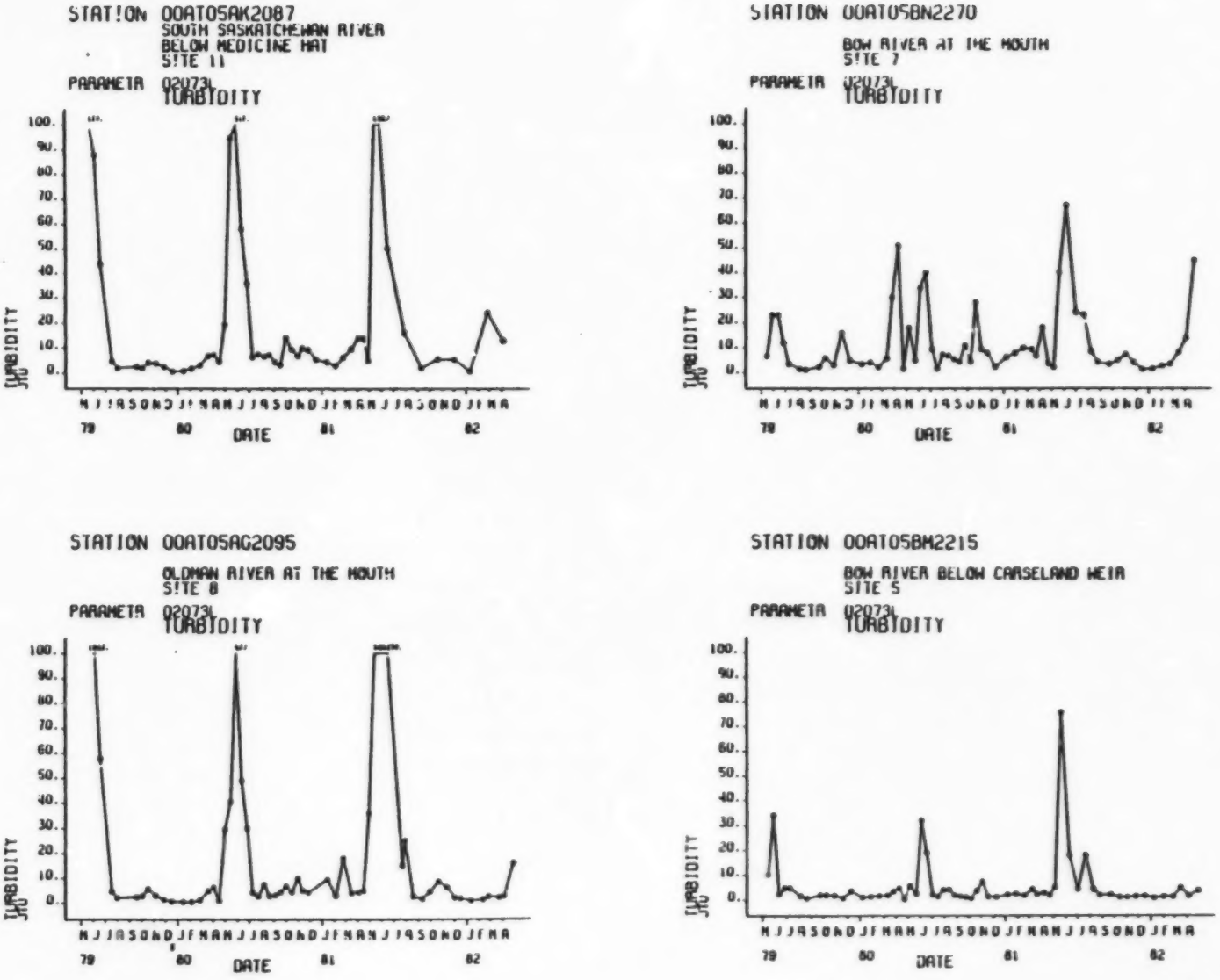


Fig. 5.3 **Comparison of turbidity data from four sites in the study basin.**

(from Cross et al., 1986)

Alberta
ENVIRONMENT
Pollution Control Division
Water Quality Control Branch

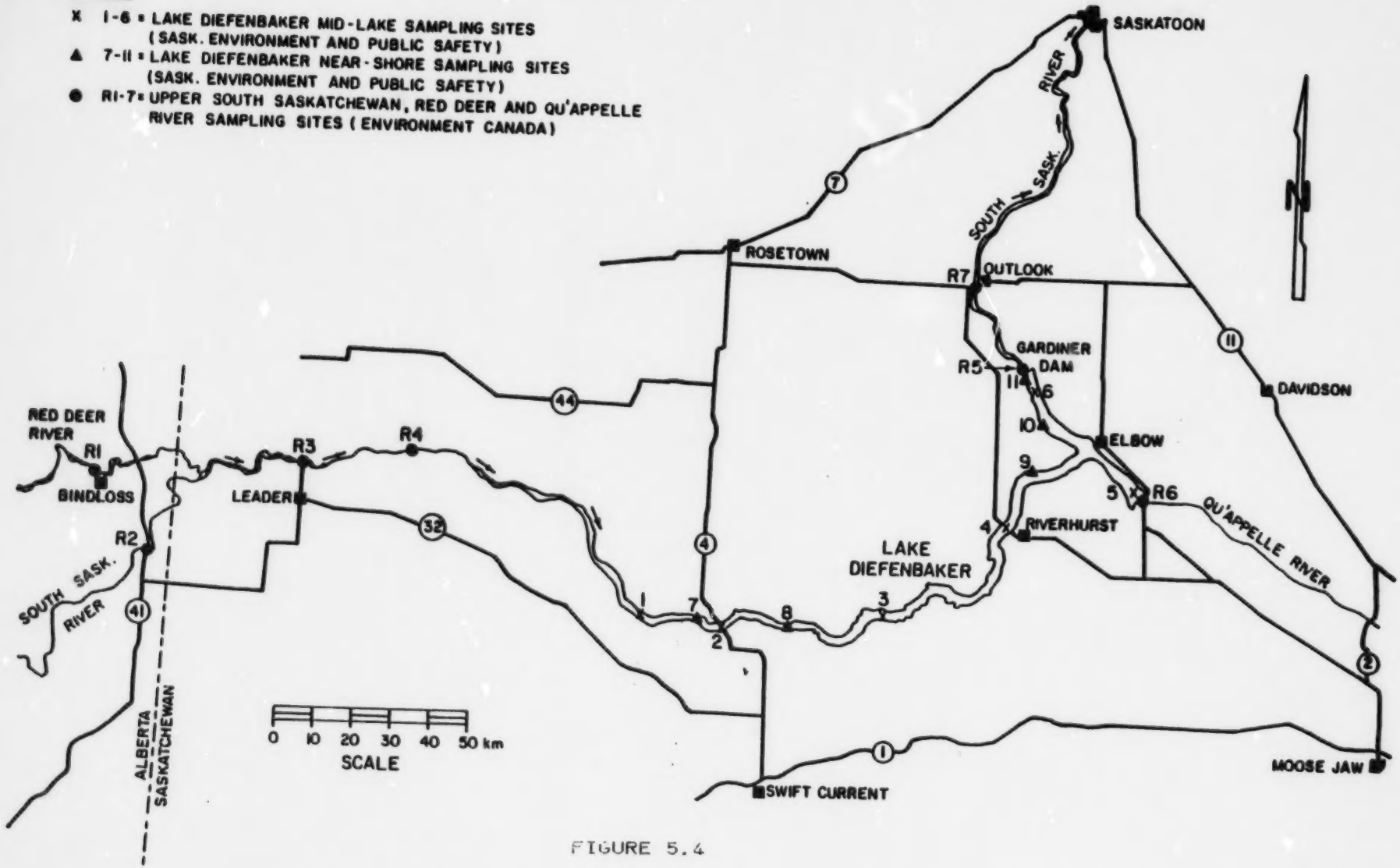


FIGURE 5.4

LAKE DIEFENBAKER AND THE UPPER SOUTH SASKATCHEWAN RIVER
STUDY SETTING AND SAMPLING SITES, 1984 - 1985

[illegible] and TWD, 1988)

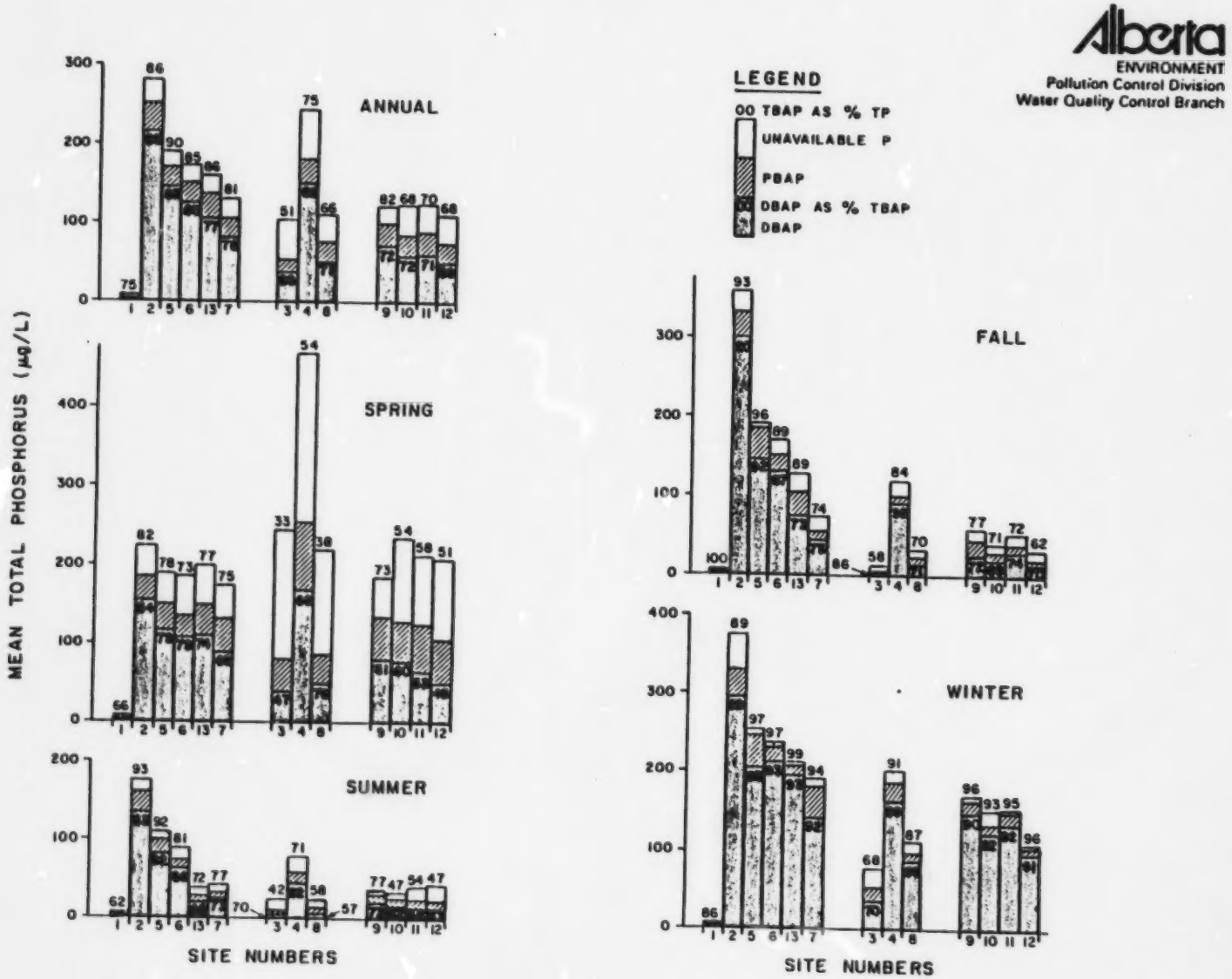


Fig. 5.5 Comparison of annual and seasonal average values for biologically available phosphorus forms and total phosphorus at each site in the study basin.

(from Cross et al., 1986)

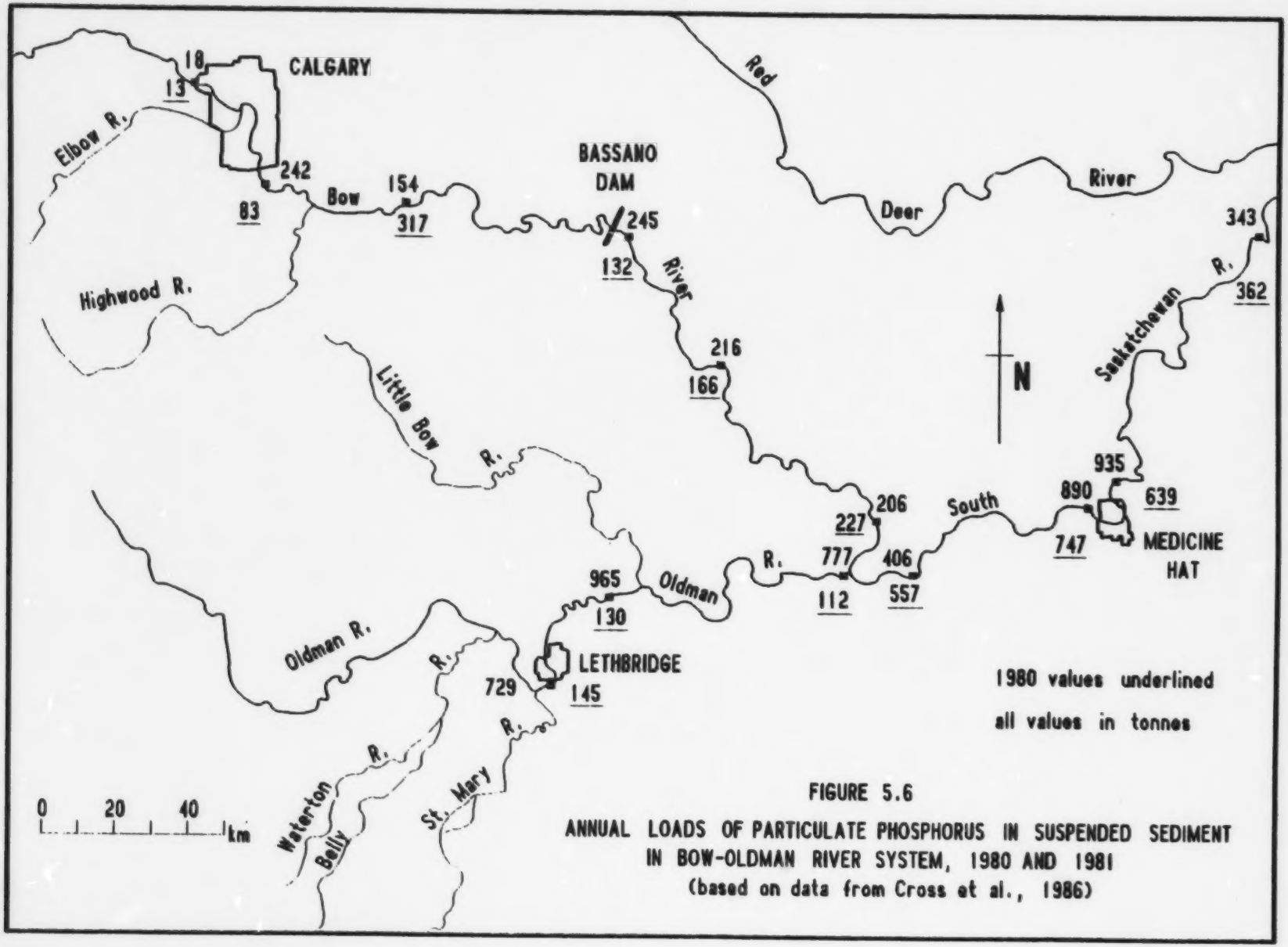


FIGURE 5.6

ANNUAL LOADS OF PARTICULATE PHOSPHORUS IN SUSPENDED SEDIMENT IN BOW-OLDMAN RIVER SYSTEM, 1980 AND 1981
(based on data from Cross et al., 1986)

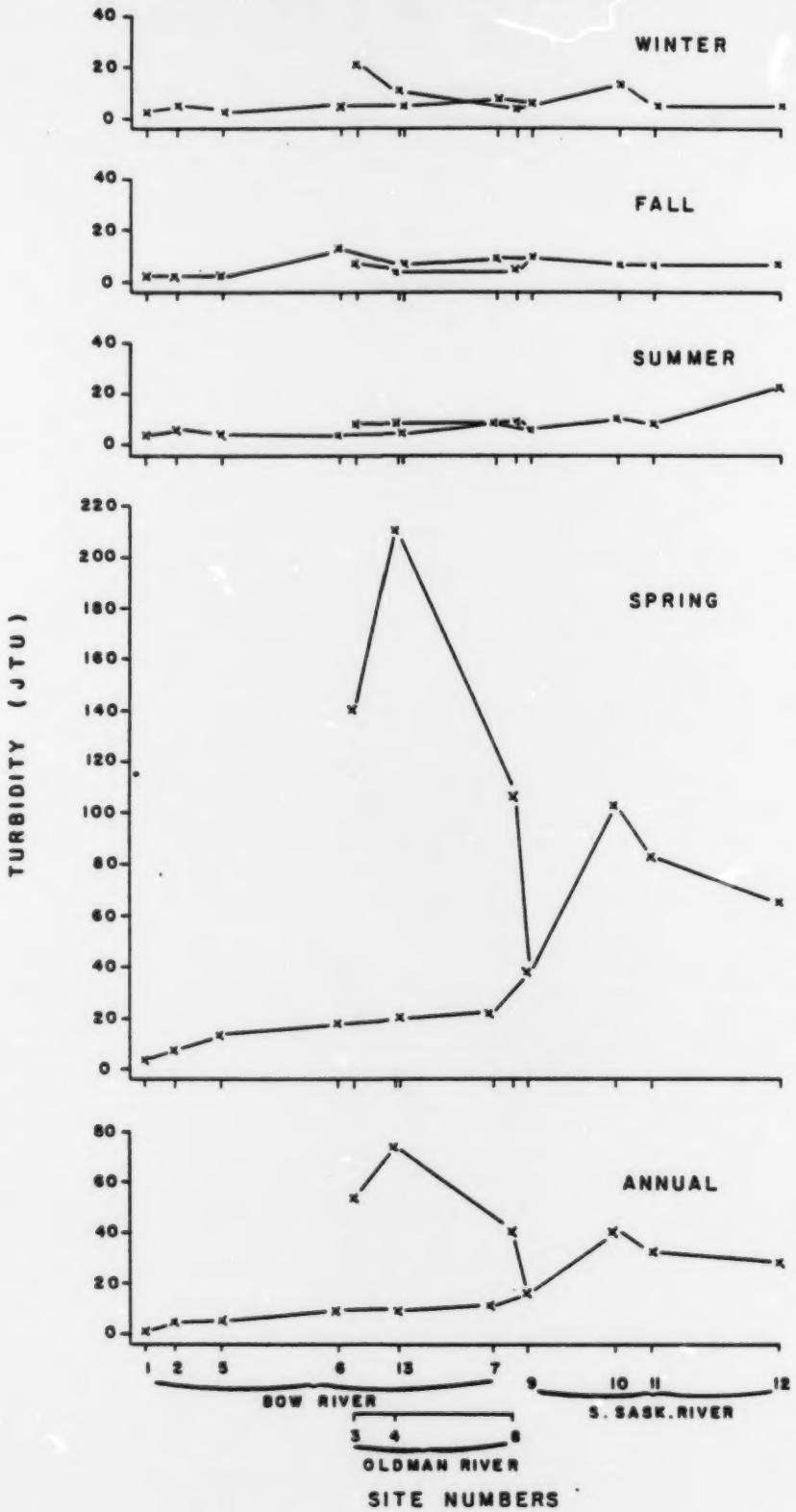


Fig. 5.7 Annual and seasonal average values of turbidity at each site in the study basin.

(from Cross et al., 1986)

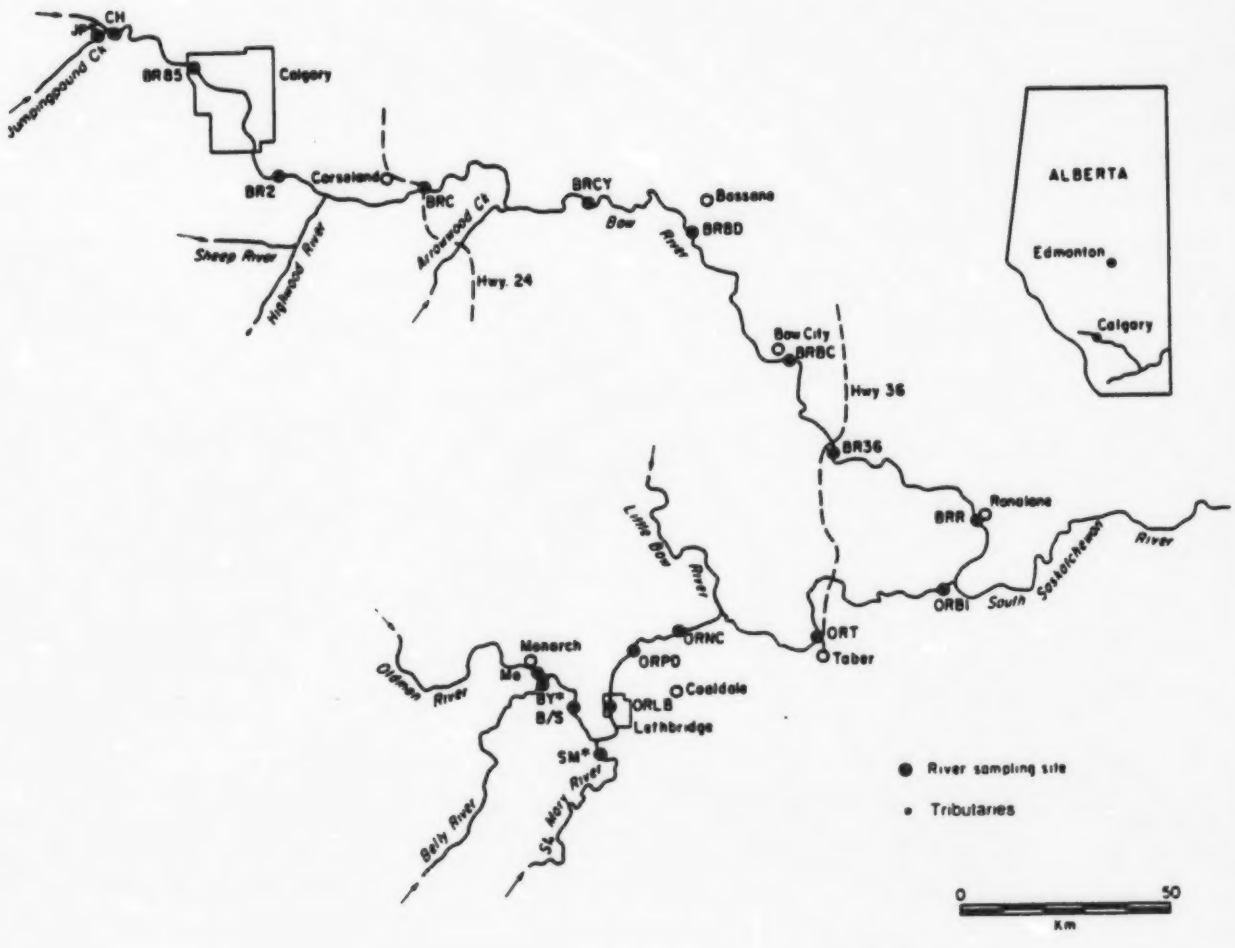


Figure 5.8

Sampling sites in the Blachford-Ongley study

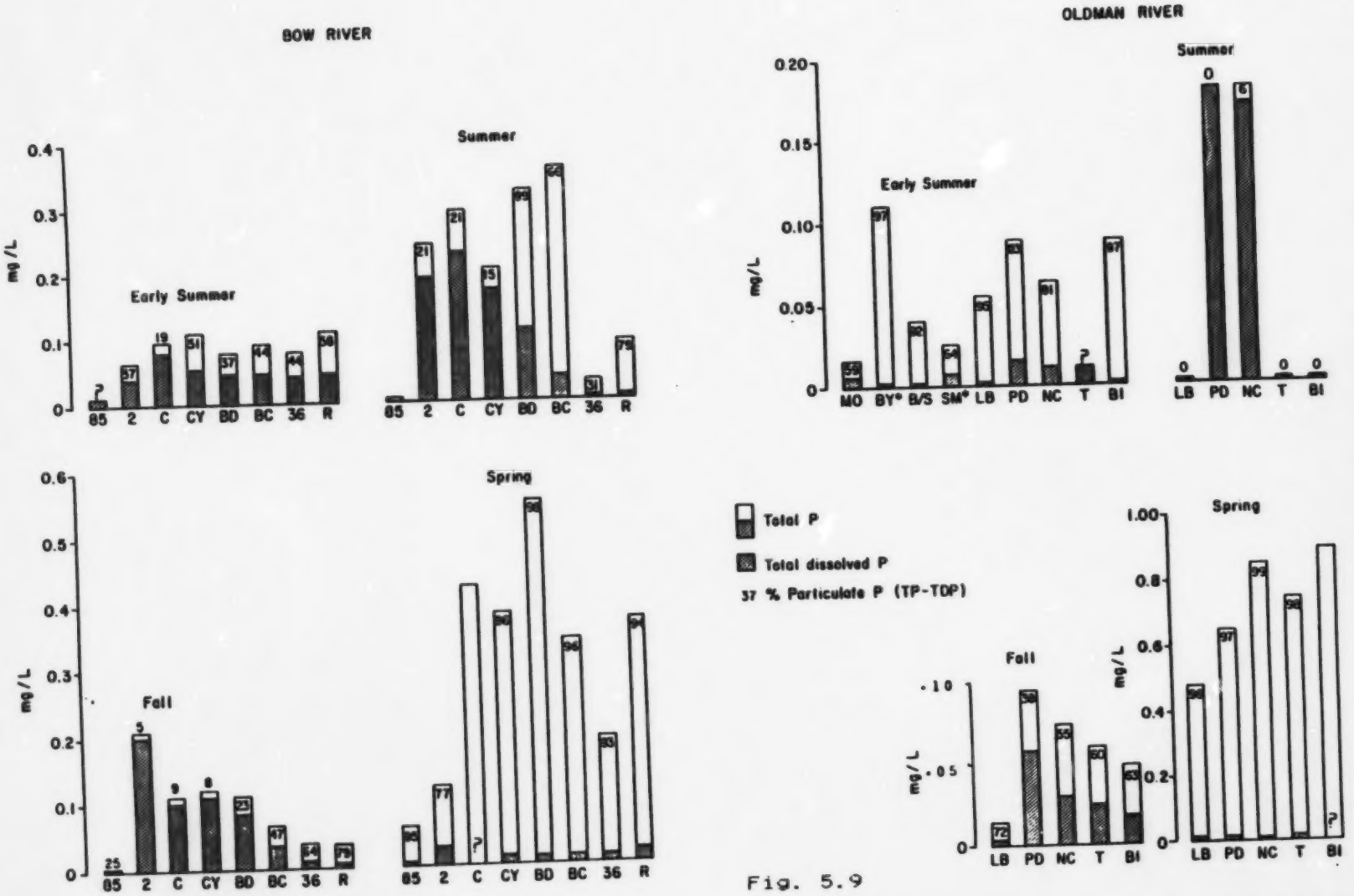
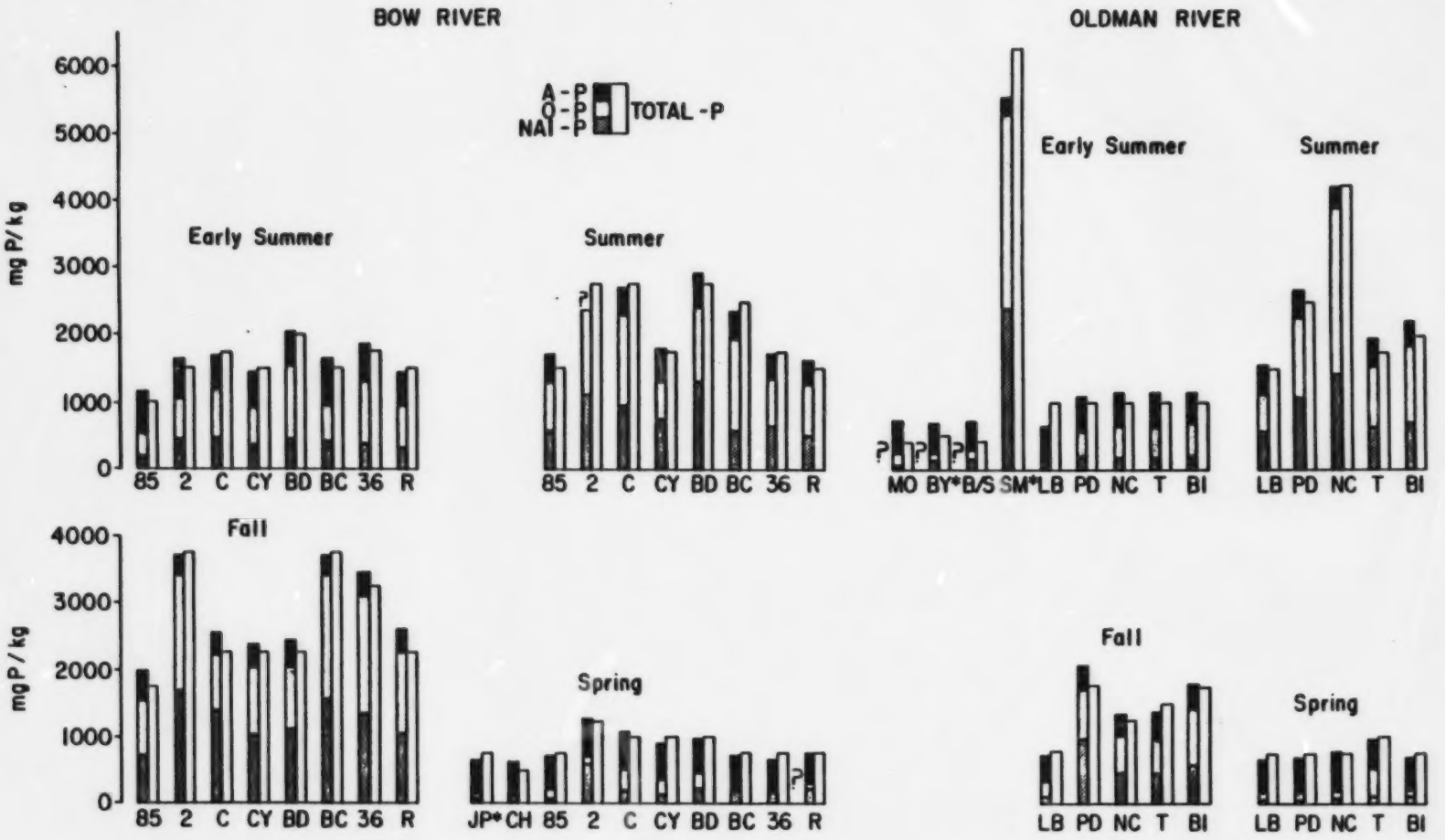


Fig. 5.9

Total and dissolved phosphorus concentrations in water. The particulate fraction is calculated as the difference between total and dissolved phosphorus. Particulate phosphorus is presented numerically on a percentage of total-P. Missing fractions are denoted by "?".

(from Blachford and Ongley, 1984)



Phosphorus speciation on suspended sediments. The amount of each phosphorus species can be calculated using the ordinate axis scale. The sum of the three species cannot be computed where one of the fractions is missing (denoted by "?"). Difference between TP and the sum of NAI-P, O-P and A-P is a measure of analytic imprecision.

Fig. 5.10 (from Blachford and Ongley, 1984)

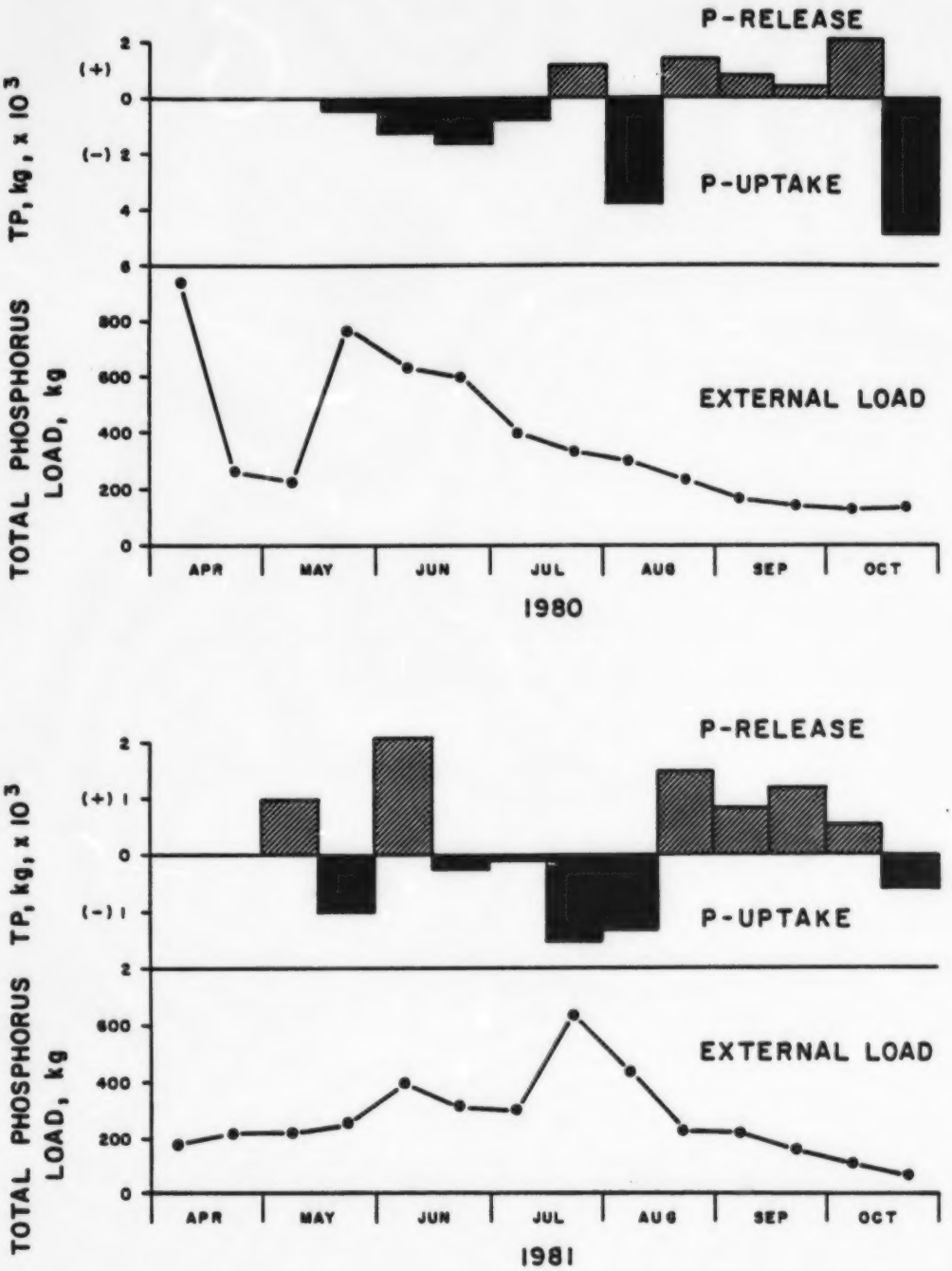


FIG. 5.11 **Sediment uptake and release of total phosphorus from mass balance calculations, and measured external load, Lake Wabamun 1980 and 1981.**

(from Mitchell, 1985)

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Pollution Control Division
Water Quality Control Branch

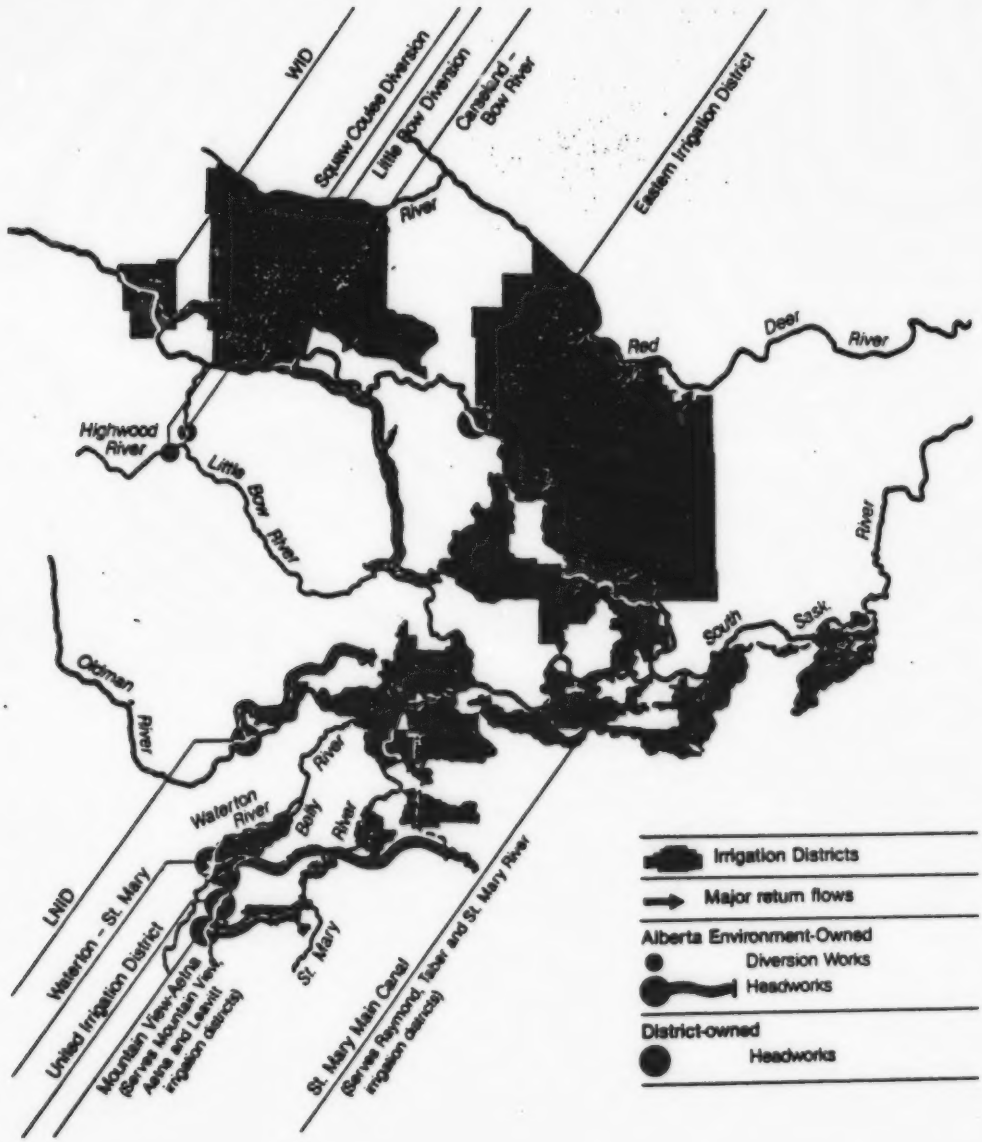
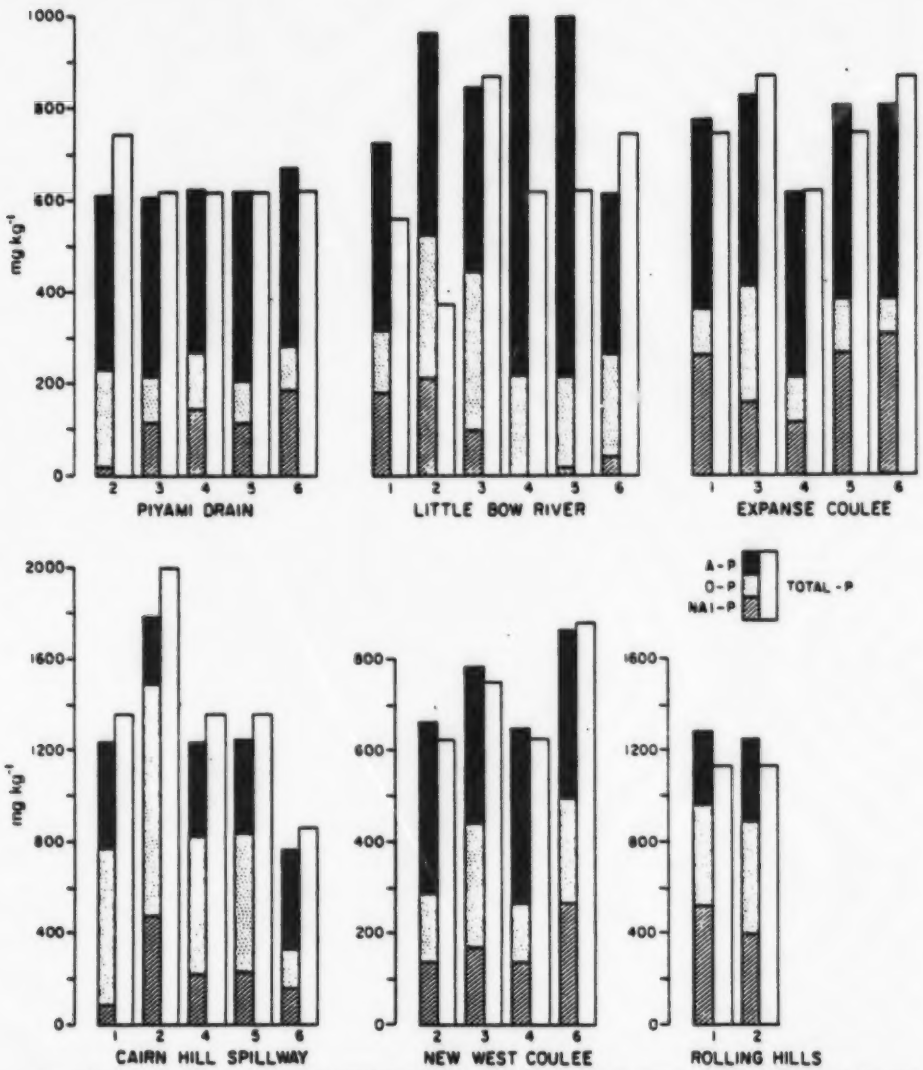


Fig. 5.12

Irrigation water supply systems in Alberta

(from PPWB. 1982)



Phosphorus species on suspended sediment from selected irrigation return flows. Difference between T-P (a separate analysis) and the sum of NAI-P, O-P, and A-P is a measure of analytical imprecision.

Figure 5.13

(Taken from Joseph and Ongley, 1986)

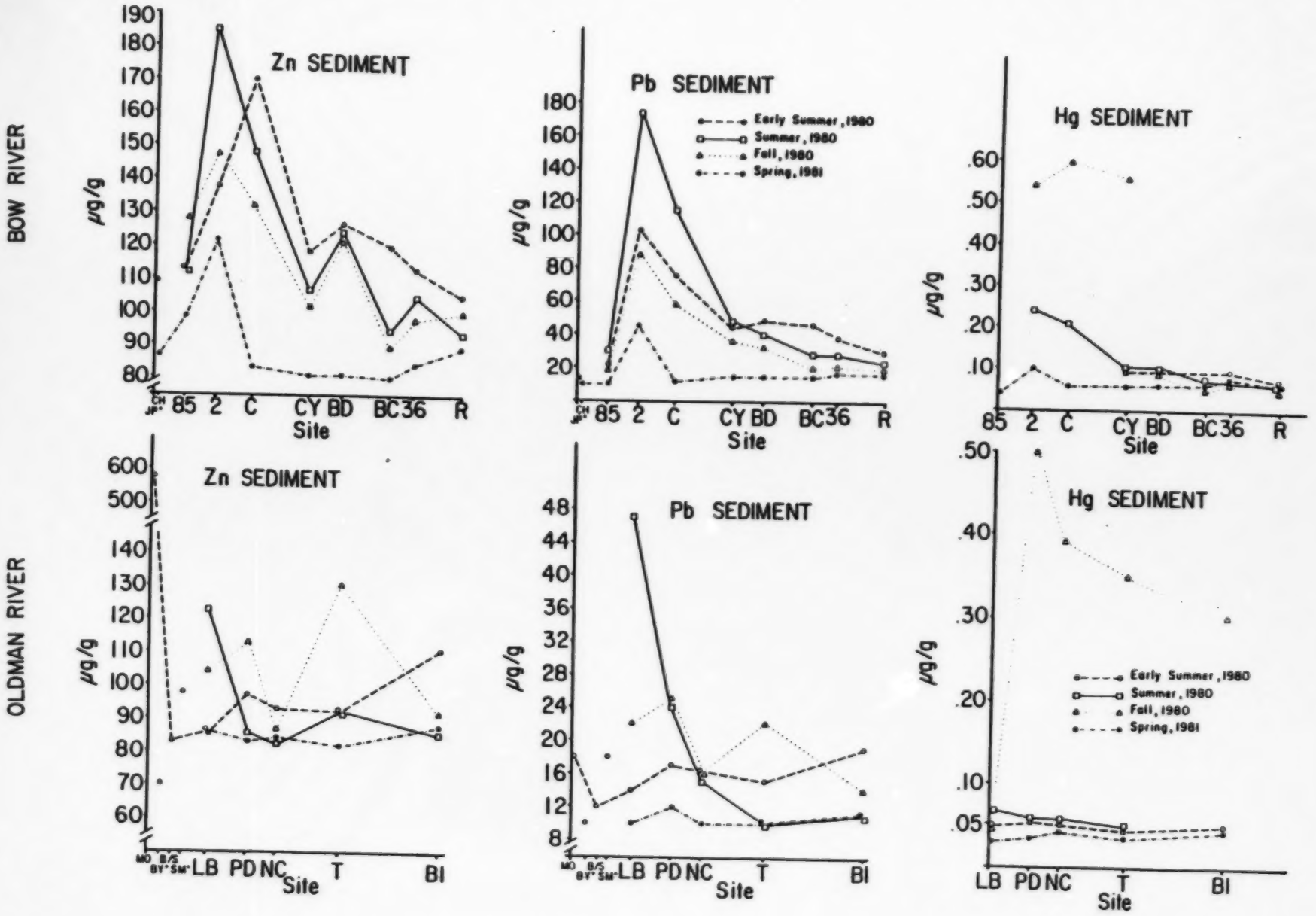


Fig. 6.1 Downstream changes in metal concentrations in suspended sediment of Bow and Oldman rivers (see Fig. 5.8 for locations) (from Blachford and Ongley, 1984)

Figure 6.2 Sampling sites on the North Saskatchewan River (from Ongley et al., 1985)

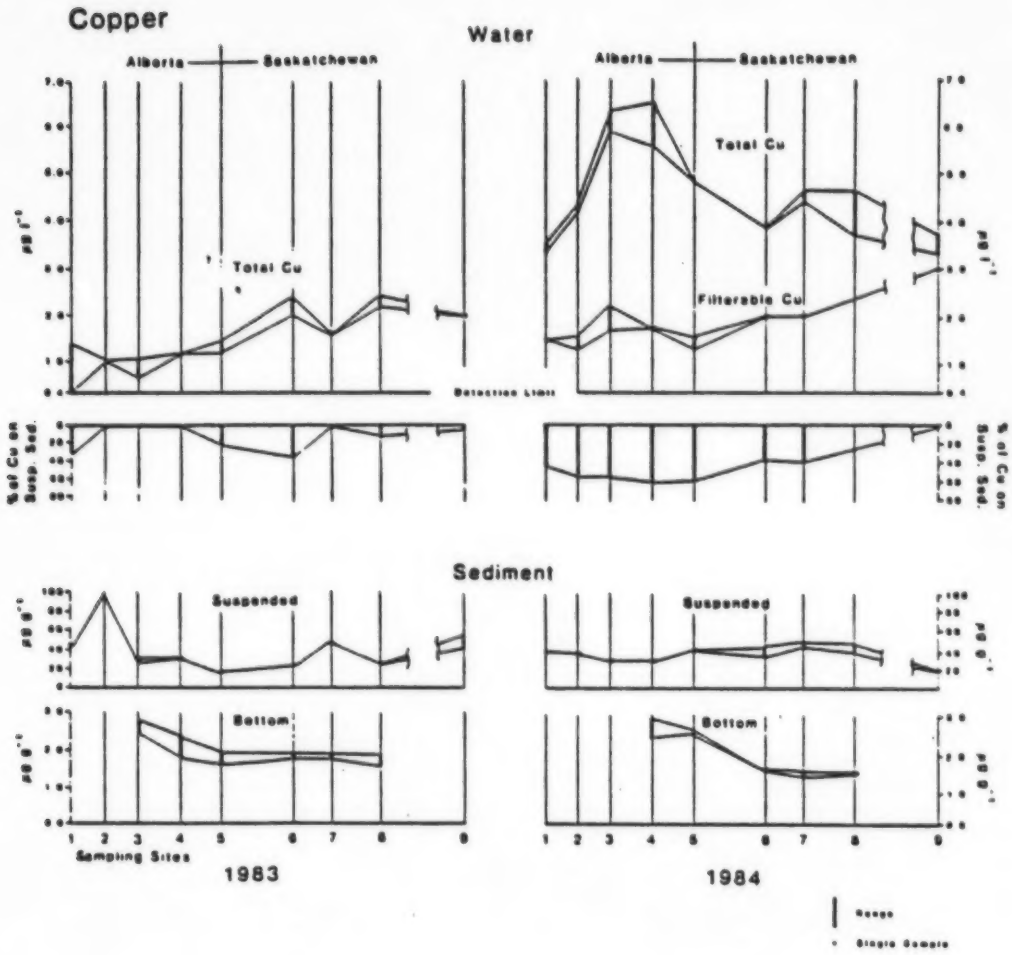


Figure 6.3

Downstream changes in copper concentrations in water, suspended sediment and bottom sediments in the North Saskatchewan River in the summer of 1983 and spring of 1984 (from Ongley et al., 1985)

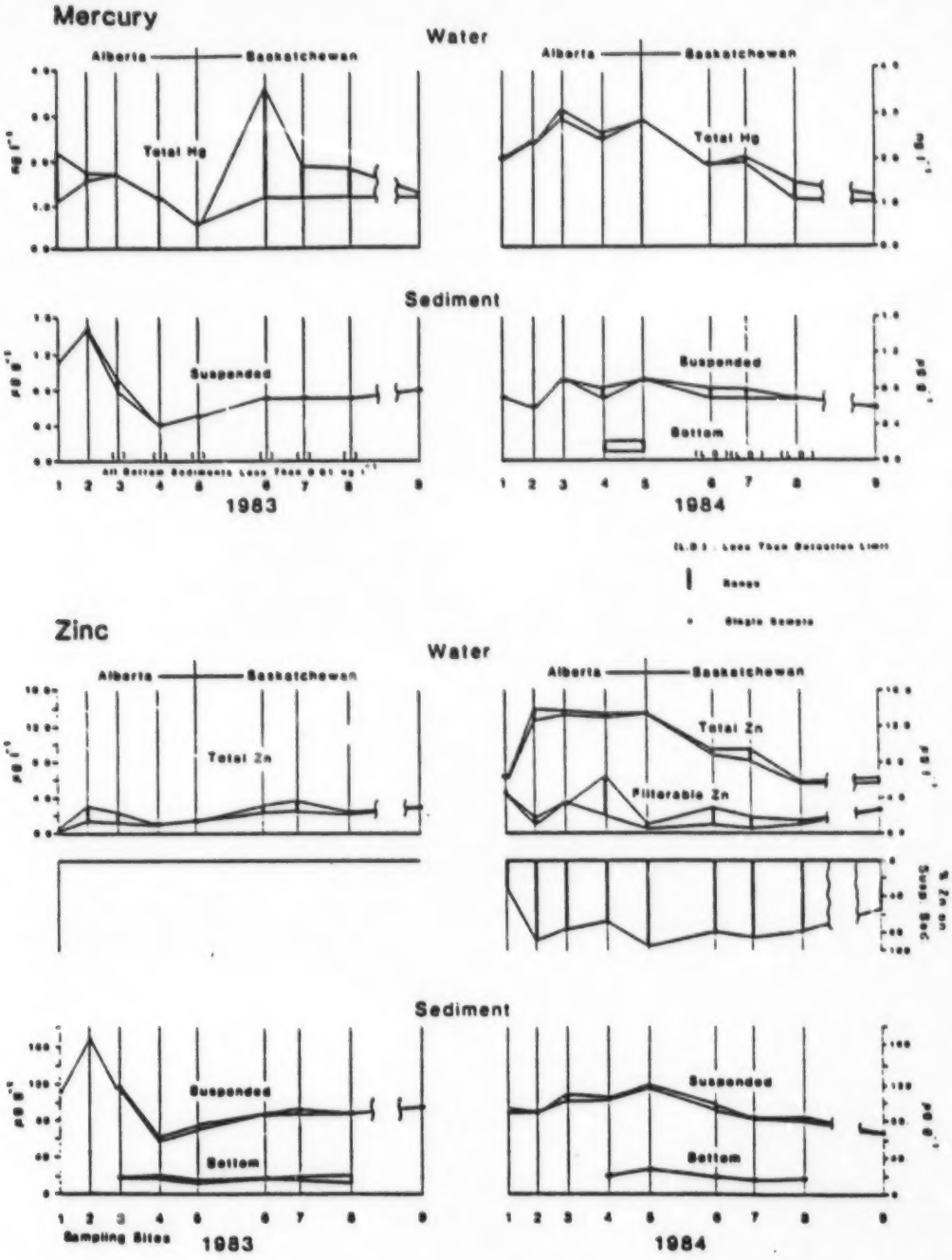
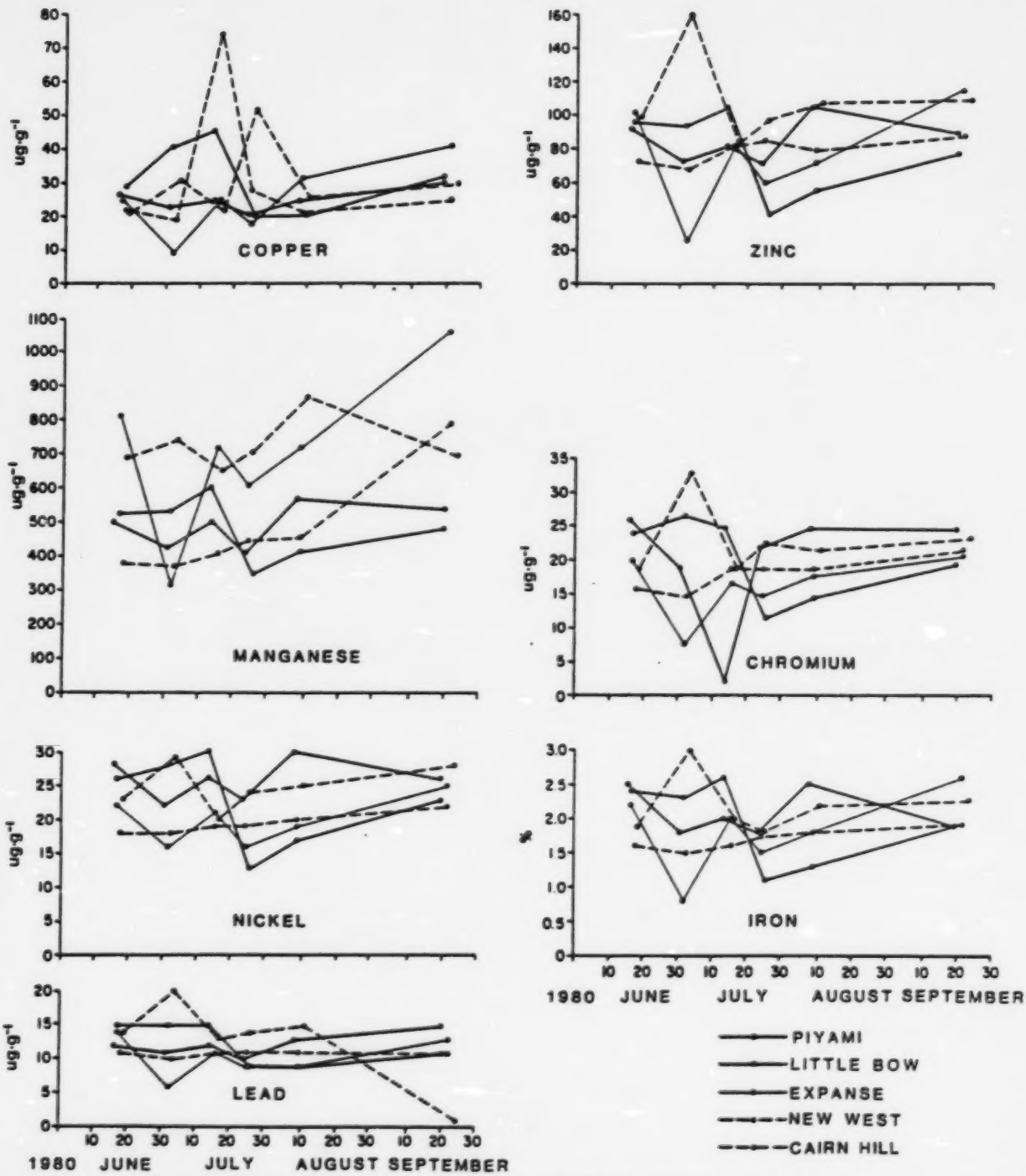


Figure 6.4

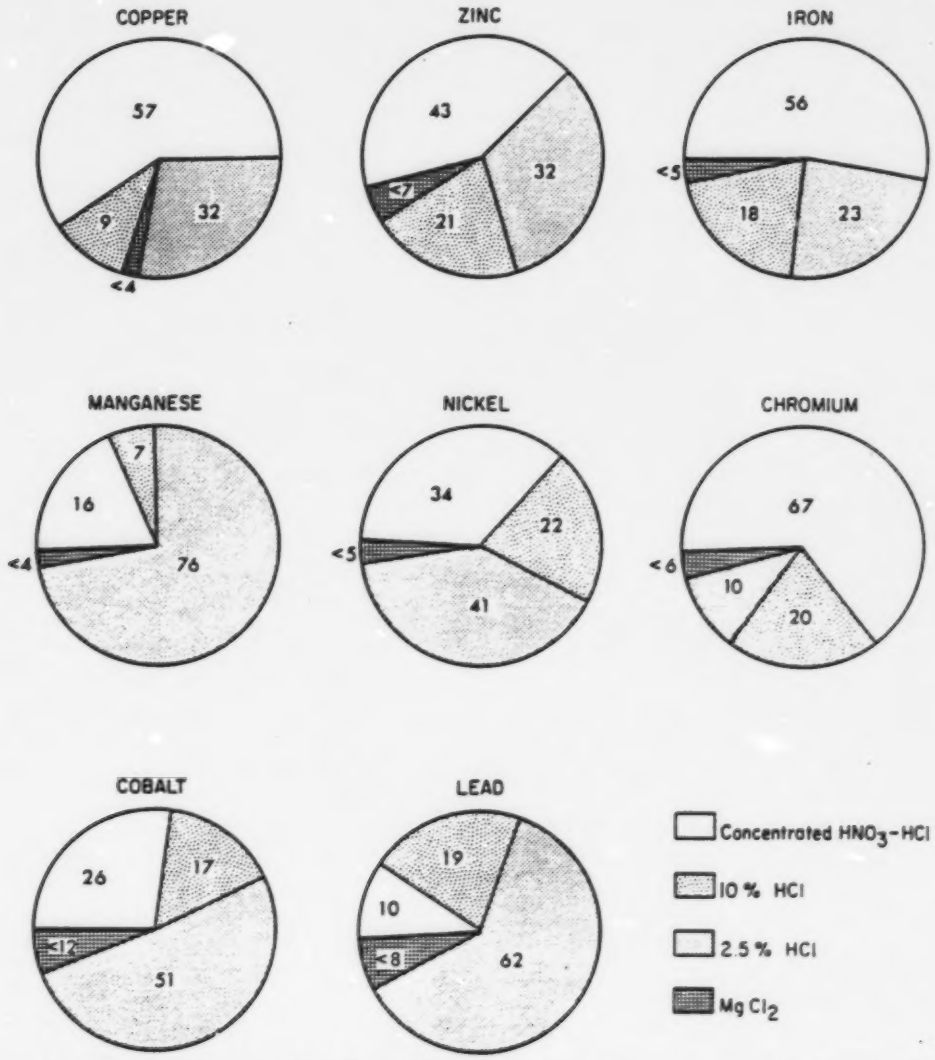
Downstream changes in concentrations of mercury and zinc along N. Saskatchewan River (from Ongley et al., 1985)



Seasonal trends of suspended sediment-associated "total" metals in sampled irrigation return flows.

Figure 6.5

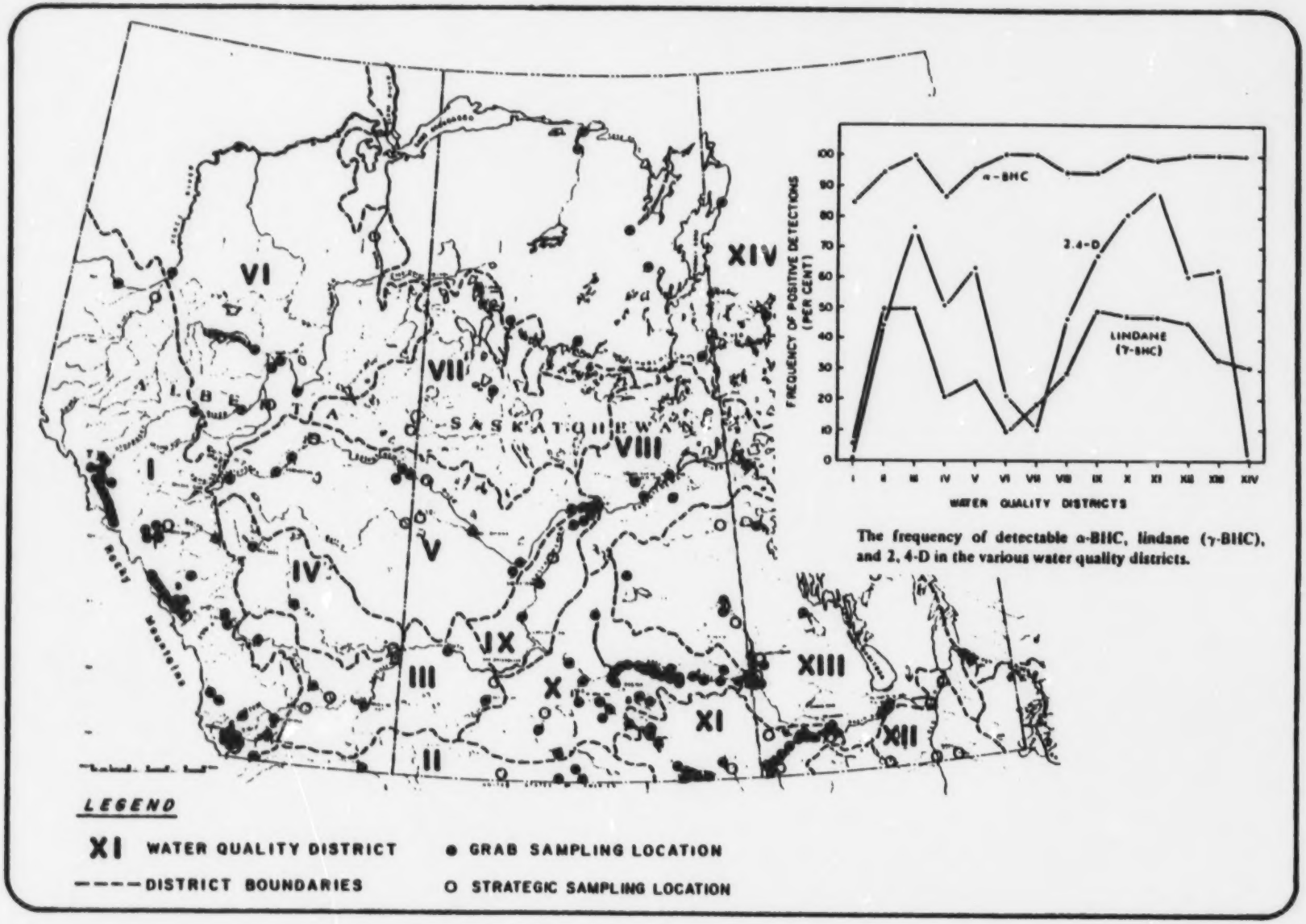
Time series for concentrations of metals in sediment sampled in irrigation return flows in southern Alberta (from Joseph and Ongley, 1986)



Proportion of metal extracted from freeze-dried sediment by each extractant expressed as a percentage of the "total" metal extracted by hot concentrated HNO_3-HCl. The concentrated HNO_3-HCl fraction is the residual amount not extracted by the other weaker extractants. Values are averages from all sites.

Figure 6.6

Speciation of metals in suspended sediment of return flows in southern Alberta (from Joseph and Ongley, 1986)



The frequency of detectable α-BHC, lindane (γ-BHC), and 2, 4-D in the various water quality districts.

FIG. 7.1 PESTICIDE LEVELS IN PRAIRIE SURFACE WATER
(from Gummer, 1979)

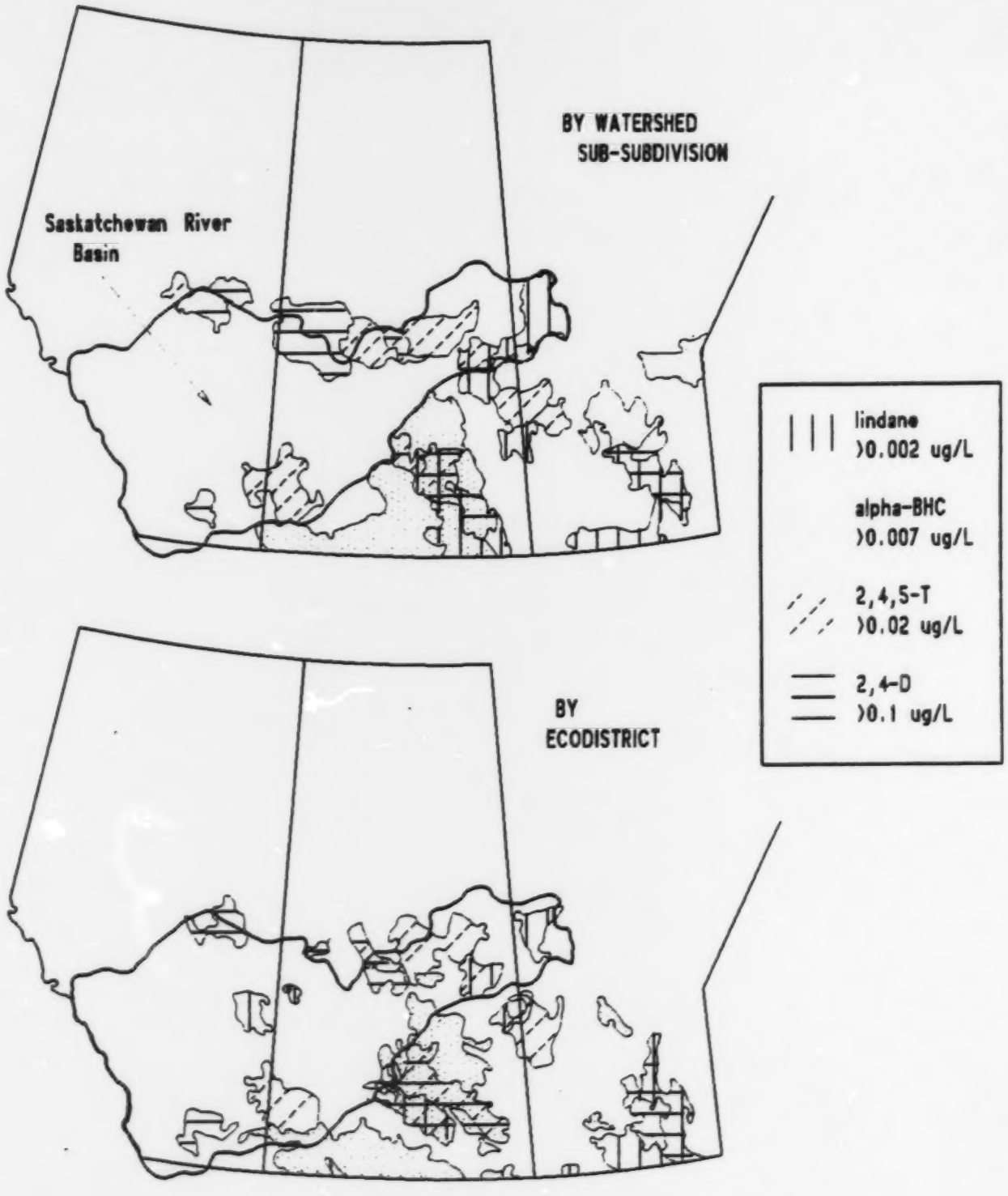
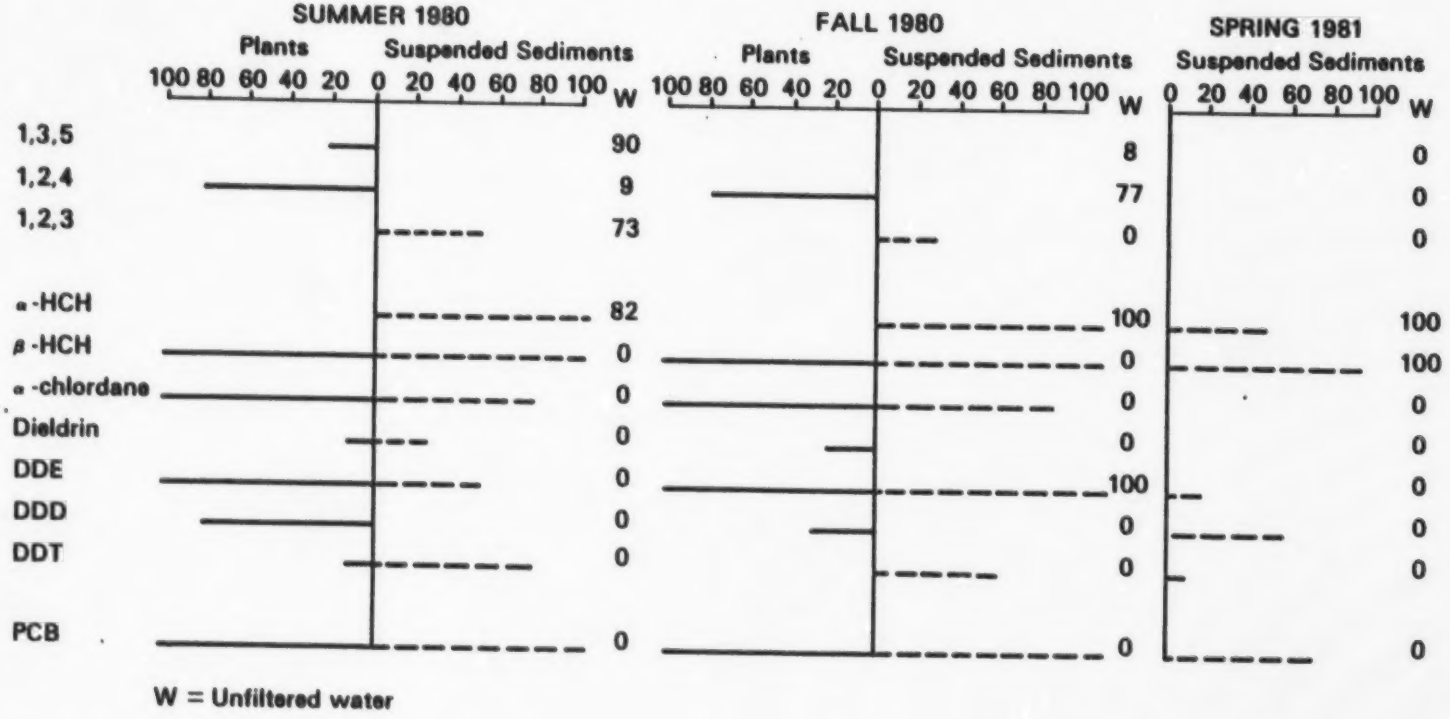


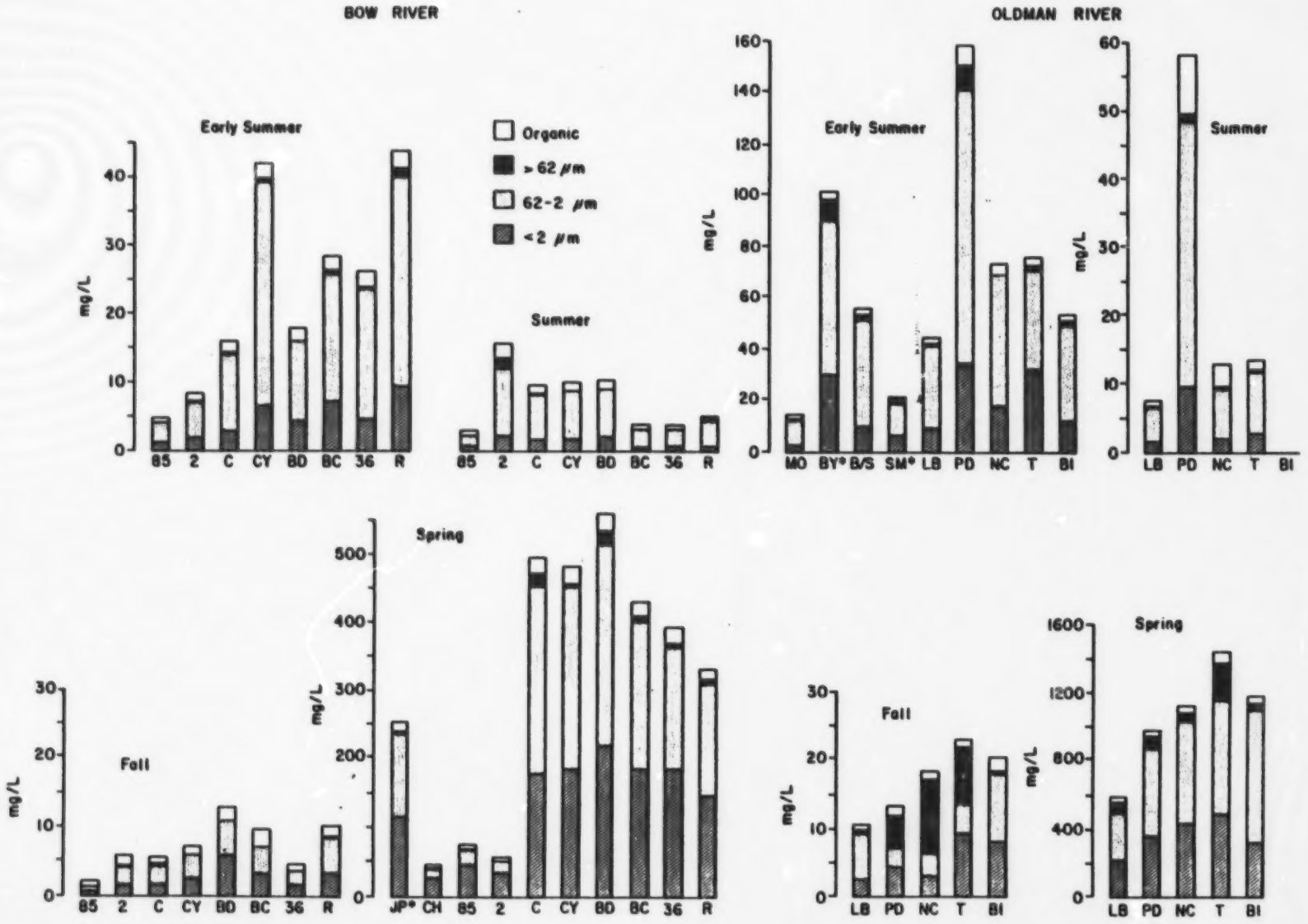
FIGURE 7.2

AREAS OF HIGH LEVELS OF PESTICIDE RESIDUES
IN SURFACE WATER OF PRAIRIE PROVINCES



Seasonal phase distributions of chlorobenzenes, organochlorines and PCBs for the Bow and Oldman rivers. The values for plants, suspended sediments and water (W) samples indicate the percentage of samples in which detectable quantities of the substances were found. Plant matter was not sufficiently available in the spring.

Figure 7.3 Presence of organic compounds in water, suspended sediment and aquatic biomass in Bow and Oldman rivers by season (from Blachford and Ongley, 1984)



Suspended sediment concentrations and organic/mineral fractions of the sediments. The height of each bar represents the total sediment concentration. The organic component and the sand (>62 μm), silt (62-2 μm) and clay (<2 μm) components of the mineral material are indicated as proportions of the total sediment concentration.

Figure 7.4 Organic and particle size composition of suspended sediment in the Bow and Oldman rivers by season